Streambank Protection with Rip-rap: An Evaluation of the Effects on Fish and Fish Habitat

J.T. Quigley and D.J. Harper
(Editors)

Habitat and Enhancement Branch
Fisheries and Oceans Canada
200 - 401 Burrard Street
Vancouver, British Columbia, V6C 3S4

2004

Canadian Manuscript Report of Fisheries and Aquatic Sciences 2701

Fisheries and Oceans Canada | Pêches et Océans Canada

Canada

Canadian Manuscript Report of Fisheries and Aquatic Sciences

Manuscript reports contain scientific and technical information that contributes to existing knowledge but which deals with national or regional problems. Distribution is restricted to institutions or individuals located in particular regions of Canada. However, no restriction is placed on subject matter, and the series reflects the broad interests and policies of the Department of Fisheries and Oceans, namely, fisheries and aquatic sciences.

Manuscript reports may be cited as full publications. The correct citation appears above the abstract of each report. Each report is abstracted in *Aquatic Sciences and Fisheries Abstracts* and indexed in the Department's annual index to scientific and technical publications.

Numbers 1-900 in this series were issued as Manuscript Reports (Biological Series) of the Biological Board of Canada, and subsequent to 1937 when the name of the Board was changed by Act of Parliament, as Manuscript Reports (Biological Series) of the Fisheries Research Board of Canada. Numbers 1426 - 1550 were issued as Department of Fisheries and the Environment, Fisheries and Marine Service Manuscript Reports. The current series name was changed with report number 1551.

Manuscript reports are produced regionally but are numbered nationally. Requests for individual reports will be filled by the issuing establishment listed on the front cover and title page. Out-of-stock reports will be supplied for a fee by commercial agents.

Rapport manuscrit canadien des sciences halieutiques et aquatiques

Les rapports manuscrits contiennent des renseignements scientifiques et techniques qui constituent une contribution aux connaissances actuelles, mais qui traitent des problèmes nationaux ou régionaux. La distribution en est limitée aux organismes et aux personnes de régions particulières du Canada. Il n'y a aucune restriction quant au sujet; de fait, la série reflète la vaste gamme des intérêts et des politiques du ministère des Pêches et des Océans, c'est-à-dire les sciences halieutiques et aquatiques.

Les rapports manuscrits peuvent être cités comme des publications complètes. Le titre exact paraît au-dessus du résumé de chaque rapport. Les rapports manuscrits sont résumés dans la revue *Résumés des sciences aquatiques et halieutiques*, et ils sont classés dans l'index annual des publications scientifiques et techniques du Ministère.

Les numéros 1 à 900 de cette série ont été publiés à titre de manuscrits (série biologique) de l'Office de biologie du Canada, et après le changement de la désignation de cet organisme par décret du Parlement, en 1937, ont été classés comme manuscrits (série biologique) de l'Office des recherches sur les pêcheries du Canada. Les numéros 901 à 1425 ont été publiés à titre de rapports manuscrits de l'Office des recherches sur les pêcheries du Canada. Les numéros 1426 à 1550 sont parus à titre de rapports manuscrits du Service des pêches et de la mer, ministère des Pêches et de l'Environnement. Le nom actuel de la série a été établi lors de la parution du numéro 1551.

Les rapports manuscrits sont produits a l'échelon régional, mais numérotés à l'échelon national. Les demandes de rapports seront satisfaites par l'établissement auteur dont le nom figure sur la couverture et la page du titre. Les rapports épuisés seront fournis contre rétribution par des agents commerciaux.

105-01513/

Canadian Manuscript Report of
Fisheries and Aquatic Sciences 2701

2004

Streambank Protection with Rip-rap: An Evaluation of the Effects on Fish and Fish Habitat

by

J.T. Quigley and D.J. Harper
(Editors)

Habitat and Enhancement Branch
Fisheries and Oceans Canada
200 - 401 Burrard Street
Vancouver, British Columbia, V6C 3S4

© Her Majesty the Queen in Right of Canada, 2004,
Cat. No. Fs 97-4/2701E ISSN 0706-6473
PDF Version: Cat. No. Fs 97-4/2701E-PDF ISSN 1488-5387

Correct citation for this publication:

Quigley, J.T. and Harper, D.J. (Editors). 2004. Streambank protection with rip-rap: an evaluation of the effects on fish and fish habitat. Can. Manuscr. Rep. Fish. Aquat. Sci. 2701: xiv + 76 p.

TABLE OF CONTENTS

TABLE OF CONTENTS iii

LIST OF TABLES v

LIST OF FIGURES viii

ABSTRACT x

RÉSUMÉ xii

1.0 INTRODUCTION 1

2.0 LITERATURE REVIEW ON THE EFFECTS OF RIP-RAP ON FISH AND FISH HABITAT WITH HABITAT MANAGEMENT IMPLICATIONS 3

_2.1 BACKGROUND 4
_2.2 PURPOSE AND APPLICATION OF RIP-RAP 4
_2.3 SCOPE OF THIS REVIEW 4
_2.4 METHODOLOGY 5
_2.5 RESULTS 5
2.5.1 Positive Effects of Rip-rap on Fish and Fish Habitat 5
2.5.2 Negative Effects of Rip-rap on Fish and Fish Habitat 8
2.5.3 Mitigation of the Negative Effects 10
2.5.4 Compensation Measures 12
2.5.5 Alternatives to Rip-rap 12
_2.6 DISCUSSION AND RECOMMENDATIONS 14
2.6.1 Management Objectives 14
2.6.2 Alternatives to Rip-rap 15
2.6.3 Research Recommendations 15
_2.7 REFERENCES 16
2.7.1 Personal Communications 16
2.7.2 Electronic Sources 16
2.7.3 Literature Reviewed 16

3.0 FIELD EVALUATION OF RIP-RAP EFFECTS ON JUVENILE SALMONID HABITAT 21

_3.1 INTRODUCTION 22
_3.2 FIELD STUDY AREAS 22
_3.3 CASE STUDIES 25
_3.4 NORTH THOMPSON RIVER 25
3.4.1 Methods 25
3.4.2 Results 29
_3.5 THOMPSON RIVER 39
3.5.1 Methods 39
3.5.2 Results 42
_3.6 COLDWATER RIVER 46
3.6.1 Methods 46
3.6.2 Results 48

_3.7 LOWER FRASER RIVER 54
3.7.1 Methods 54
3.7.2 Results 58
_3.8 DISCUSSION 67
3.8.1 Species, Life Stage and Season 67
3.8.2 Stream Size and Hydrology 68
3.8.3 Rip-rap Character 68
3.8.4 Vegetation at Rip-rap Banks 69
_3.9 ACKNOWLEDGEMENTS 69
_3.10 REFERENCES 70
4.0 CONCLUSIONS AND RECOMMENDATIONS 73
_4.1 CONCLUSIONS 73
_4.2 RECOMMENDATIONS 74
4.2.1 Short term 74
4.2.2 Long term 74
_4.3 REFERENCES 76

LIST OF TABLES

No.		Page
1.	Distribution of North Thompson River fish sampling sites among bank and channel types by river section and season.	28
2.	Age composition and mean fork length at age of juvenile coho in upper and lower river study areas.	30
3.	Mean values for juvenile coho density and selected habitat parameters, organized by bank and channel type.	32
4.	Percentages of natural and rip-rap bank sampling sites with selected characteristics, North Thompson River, late summer.	34
5.	Regression model predictions of juvenile coho rearing densities at natural and rip-rap banks over a range of nearshore velocities, North Thompson River, late summer.	35
6.	Mean values of coho density, nearshore velocity and riparian vegetation at selected natural and rip-rap sampling sites with relatively favourable vegetation (> 30% of site length) and velocity (< 25 cm/s) conditions, North Thompson River, late summer.	37
7.	Mean densities of juvenile chinook and coho at natural and rip-rap banks, North Thompson River in winter.	38
8.	Estimated bank D90 at natural and rip-rap bank sampling sites, North Thompson River.	38
9.	Numbers of paired comparisons of juvenile salmonid densities by bank type, Thompson River, 1987.	39
10.	Nearshore water velocity criteria used in habitat classification for paired site comparisons, Thompson River, 1987.	41
11.	Mean fork lengths of juvenile chinook subyearlings and rainbow trout parr (age 1+ and 2+) in the Thompson River during summer.	42

12.	Estimated mean rock size at Thompson River study sites at high flow in June and medium flow in September.	42
13.	Mean chinook subyearling and rainbow trout parr densities at paired large rip-rap and natural cobble-boulder banks, Thompson River, May 21 - June 10.	43
14.	Mean juvenile chinook and rainbow trout parr densities and water depth and velocity at natural, cobble-boulder and rip-rap banks, Thompson River, late winter.	45
15.	Physical characteristics of paired natural and rip-rap bank sampling units, Coldwater River mainstem, late winter, 1989.	48
16.	Mean fork length and length range of juvenile salmonids by sampling season, Coldwater River mainstem, 1987-1988.	49
17.	Average discharge, temperature and water level of the lower Fraser River during field surveys in 1985 and 1986.	54
18.	Results of tests to estimate electrofishing capture efficiency of chinook juveniles at lower Fraser River sampling sites during spring and summer.	57
19.	Mean length and length range of chinook subyearlings by sampling period, lower Fraser River, 1985.	58
20.	Mean values of juvenile chinook density and selected habitat parameters at natural and rip-rap bank sampling sites during spring and summer surveys, lower Fraser River.	60
21.	Mean chinook density, vegetation and nearshore velocity at high and low cover sites by bank type in June and July, lower Fraser River.	62
22.	Seasonal change in relative abundance of juvenile chinook at high and low velocity sites, lower Fraser River, June – August.	63
23.	Fish sampling effort and numbers of natural and rip-rap bank sampling sites that supported over-wintering salmonids, lower Fraser River.	65

24. Mean values of juvenile salmonid density and measured habitat parameters at natural and rip-rap bank sampling sites in each study section, lower Fraser River, late winter. 66

LIST OF FIGURES

No.		Page
1.	Study area locations in the lower Fraser River Basin.	24
2.	Natural bank (a) and rip-rap bank (b) study sites, North Thompson River, late summer.	27
3.	Scatterplot matrix showing relationships among juvenile coho density (COHO) and habitat variables, including bank material size (D90), nearshore water velocity (AVEL) and depth (DEPTH), bank vegetation (VEG) and wood debris (DEBRIS) at North Thompson River study sites in late summer.	31
4.	Juvenile coho density relative to nearshore water velocity, riparian vegetation and wood debris at natural and rip-rap bank sites, North Thompson River, late summer.	33
5.	Juvenile coho density relative to nearshore velocity and riparian vegetation at natural (NAT) and rip-rap (RR) banks, North Thompson River, late summer.	36
6.	Natural cobble-boulder bank (a), small rip-rap bank (b) and large rip-rap bank (c) study sites, Thompson River at low flow in April.	40
7.	Mean juvenile chiook subyearling (a) and rainbow trout parr (b) densities at large rip-rap (LR), small rip-rap (SR) and cobble-boulder (CB) banks, Thompson River.	44
8.	Natural banks with high cover (a) and low cover (b) rating, and rip-rap bank (c), Coldwater River, late summer.	47
9.	Mean juvenile salmonid densities at paired altered and reference sites, Coldwater River mainstem, late summer.	50
10.	Mean juvenile chinook and coho densities at paired natural and rip-rap bank sites relative to water depth (deep and shallow) and high (HC) or low (LC) bank cover, Coldwater River mainstem, late winter.	51
11.	Mean juvenile steelhead 0+ and parr densities at paired natural and rip-rap bank sites relative to water depth (deep and shallow) and high (HC) or low (LC) bank cover, Coldwater River mainstem, late winter.	52

12. Mean mountain whitefish and bull trout densities at Coldwater River mainstem altered and reference sites. 53

13. Natural bank in August (a) and vegetated rip-rap site at high flow in June (b) and at lower flow in August (c), lower Fraser River. 55

14. Juvenile chinook density at natural and rip-rap banks in relation to selected habitat parameters during May (a) and June (b), lower Fraser River. 61

15. Juvenile chinook density at natural and rip-rap banks in relation to selected habitat parameters in July (a) and August (b), lower Fraser River. 64

ABSTRACT

Quigley, J.T. and Harper, D.J. (Editors). 2004. Streambank protection with rip-rap: an evaluation of the effects on fish and fish habitat. Can. Manuscr. Rep. Fish. Aquat. Sci. 2701: xiv + 76 p.

This study provides habitat management staff in the Pacific Region of Fisheries and Oceans Canada (DFO) with the most recent information on the range of potential effects of rip-rap on fish and fish habitat in order to improve decision making and regional consistency. The effects of rip-rap on fish and fish habitat were investigated through a literature review and an evaluation of over 20 years of environmental impact monitoring data in British Columbia.

At a local scale, positive and negative effects of rip-rap on fish and fish habitat were found. Positive effects tended to occur with applications of rip-rap in degraded systems that had excessive erosion problems, particularly where it was combined with other habitat management strategies. Negative effects of rip-rap applications were related to the loss of natural habitat, as vegetated natural banks almost universally provided superior habitat than rip-rapped sites. In fact, our study suggested that sites with only 14% of their site length vegetated provided higher habitat value than rip-rapped sites.

Watershed scale effects include restricted lateral channel migration, decreased natural sediment deposition, reduced recruitment of gravel and large woody debris, hydrological changes (such as reduced ability to attenuate flood peaks) and reduced heterogeneity of habitat. These effects are largely negative and potentially cumulative in nature.

The effects from rip-rap should not be considered solely on a site specific basis. Impacts to landscape-level ecological and hydrological processes should be considered during the assessment of proposed rip-rap projects.

Short term management recommendations:

1. DFO should be precautionary in the use of rip-rap, and only consider it as an option in degraded systems that are devoid of riparian vegetation and subject to excessive erosion.

2. DFO should encourage the use of alternative streambank stabilization methods, such as bio-engineering approaches (Adams 2003; MELP 2000), that may provide a greater value to fish habitat.

3. The application of rip-rap at vegetated sites will likely result in a loss of productive capacity and should be Authorized pursuant to Section 35(2) of the *Fisheries Act* provided the impacts from the proposed development cannot be fully mitigated.

4. If rip-rap is the only viable alternative for streambank protection, and is to be applied at a scale large enough to potentially affect river processes, DFO should require a hydrological assessment. The impact assessment should consider landscape-level ecological and hydrological processes, and downstream cumulative impacts to the watershed.

5. For Authorized rip-rap streambank protection projects, DFO should require compensation, including additional techniques other than simply re-vegetating the site. Consideration should be given to off-channel habitat development to compensate for the loss of potential for natural off-channel habitat development. Rigorous monitoring and maintenance programs should be implemented to ensure compensation success.

Long-term management recommendations:

6. Dialogue with other levels of government and industry groups responsible for linear development (pipelines, railway, highways, etc.) should be initiated and maintained to develop strategies for protection of watersheds from excessive application of rip-rap as a bank protection technique.

7. Long term plans for river migration corridors should be developed. These plans should incorporate flood control solutions as alternatives to channelization. Examples include setback dykes at anticipated problem areas, stepped levees, greenbelts, meander zones, riparian conservation easements, subdivision regulations, building codes and zoning ordinances. A pro-active approach to planning can increase the range of solutions available, and avoid dealing with projects on an emergency basis where potential solutions can be limited.

8. A coordinated education and awareness initiative should be undertaken targeting all private and industrial landowners and stakeholders within and adjacent to river migration corridors. The value of natural riparian vegetation in terms of both stream bank protection and fish habitat should be promoted to reduce unnecessary riparian removal and subsequent activities that result in an overall hardening of stream banks within a watershed without due consideration of its overall impact.

9. The effectiveness of alternate bio-engineering approaches to streambank protection should be quantified using a scientific monitoring and assessment program.

10. Long term field research should be conducted to fill the information gap on cumulative watershed level impacts resulting from extensive rip-rapping.

RÉSUMÉ

Quigley, J.T. and Harper, D.J. (Editors).2004. Streambank protection with rip-rap: an evaluation of the effects on fish and fish habitat. Can. Manuscr. Rep. Fish. Aquat. Sci. 2701: xiv + 76 p.

Cette étude offre aux spécialistes de la gestion de l'habitat dans la Région du Pacifique de Pêches et Océans Canada (MPO) les plus récentes informations sur la gamme des effets potentiels des perrés sur les poissons et leurs habitats afin d'améliorer les processus de prise de décisions et d'assurer une certaine cohérence au niveau régional. Les effets des perrés sur les poissons et sur l'habitat du poisson ont été étudiés en effectuant une analyse documentaire et une évaluation des données accumulées sur plus de vingt ans en Colombie-Britannique dans le cadre de la surveillance des impacts environnementaux.

À l'échelle locale, ces études ont montré que les perrés pouvaient avoir des effets négatifs sur les poissons et leurs habitats. De tels ouvrages peuvent avoir des effets positifs lorsqu'ils sont installés dans des systèmes dégradés affectés par de sérieux problèmes d'érosion, en particulier lorsqu'ils sont combinés à d'autres stratégies de gestion de l'habitat. Les effets négatifs de l'installation de perrés découlent le plus souvent de la perte d'habitats naturels, les berges végétalisées offrant presque toujours un habitat d'une meilleure qualité que celle des perrés. En fait, notre étude a montré qu'il suffit que la berge d'un site soit végétalisée sur seulement 14 % de sa longueur pour qu'elle offre un meilleur habitat que si elle était équipée d'un perré sur toute sa longueur.

Les effets, à l'échelle du bassin hydrographique, incluent la restriction des chenaux latéraux de migration, la diminution de la sédimentation naturelle, la diminution du renouvellement des graviers et des gros débris ligneux, des changements hydrologiques (tels qu'une capacité moindre du lit de la rivière d'atténuer les maxima atteints lors des crues) et une réduction de l'hétérogénéité de l'habitat. Ces effets sont très négatifs et, de par leur nature, peuvent s'accumuler.

L'impact des perrés ne doit pas être considéré seulement dans le contexte d'un site particulier. Les effets sur les processus écologiques et hydrologiques à l'échelle du paysage doivent être pris en considération lors de l'évaluation des projets d'installation des perrés.

Recommandation pour la gestion à court terme :

1. MPO doit prendre des précautions lors de la mise en place des perrés et n'envisager leur installation que dans le cas des systèmes dégradés qui ne possèdent aucune végétation riparienne et qui sont l'objet d'une érosion excessive.

2. MPO devrait encourager l'utilisation d'autres méthodes de stabilisation des berges, telles que les approches faisant appel à la bio-ingénierie (Adams 2003; MELP 2000), qui peuvent être plus bénéfiques à l'habitat du poisson.

3. La mise en place de perrés sur les sites végétalisés entraîne la plupart du temps une perte de capacité de production et devrait être autorisée en vertu de l'Article 35(2) de

la *Loi sur les pêches* que si les impacts du développement proposés ne peuvent être complètement compensés.

4. Si un perré est la seule solution envisageable pour la protection d'une berge et qu'il doit couvrir une longueur telle qu'il pourrait affecter les processus liés à la rivière, MPO doit effectuer une évaluation hydrologique préalable. L'évaluation des impacts potentiels doit être axée sur les processus écologiques et hydrologiques au niveau du paysage ainsi que sur les impacts cumulatifs infligés au bassin hydrographique en aval de l'installation proposée.

5. MPO doit exiger la mise en œuvre de mesures de compensation pour tout projet autorisé mettant en jeu l'installation d'un perré pour la protection des berges. Ces mesures devrait aller plus loin que la simple revégétalisation du site. Il est nécessaire de faire particulièrement attention aux développements qui ont lieu en dehors du lit principal et d'envisager les compensations nécessaires en cas de perte d'habitat naturel dans ces secteurs. Des programmes de surveillance et d'entretien rigoureux doivent être mis en place pour faire en sorte que les mesures de compensation atteignent les objectifs souhaités.

Recommandations pour la gestion à long terme :

6. Des discussions avec les autres niveaux du gouvernement et les industries responsables de la mise en place d'installations linéaires (pipelines, rails, routes, etc.) doivent être lancées pour élaborer des stratégies visant à protéger les bassins hydrographiques contre l'utilisation excessive des perrés comme technique de protection des berges.

7. Il est nécessaire d'élaborer des plans à long terme pour les corridors de migration que constituent certaines rivières. Ces plans doivent incorporer des solutions de contrôle des inondations autres que la canalisation et faisant par exemple intervenir la mise en place de digues de retrait dans les secteurs critiques, de digues relevées, de ceintures de verdure, de zones comportant des méandres, de servitudes axées sur la conservation des zones riparienes, de règlements portant sur les lotissements, de codes de construction et d'ordonnance de zonage. Une approche proactive de la planification peut donner accès à une gamme plus vaste de solutions et éviter les situations d'urgence pour lesquelles la marge de manœuvre est beaucoup plus restreinte.

8. Il faudrait lancer une initiative de sensibilisation auprès de tous les propriétaires fonciers et tous les acteurs privés et industriels à l'intérieur et à proximité des corridors de migration que constituent les rivières. Il faut promouvoir la valeur de la végétation riparienne naturelle pour la protection des berges et l'habitat du poisson afin de réduire l'élimination de celle-ci et les activités subséquentes qui entraînent un durcissement général des berges dans les bassins hydrographiques sans prendre en compte les impacts globaux.

9. L'efficacité des approches de bio-ingénierie axées sur la protection des berges des cours d'eau devrait être mesurée en utilisant un programme scientifique de surveillance et d'évaluation.

10. Des travaux de recherche devraient être entrepris pour enquêter sur les impacts cumulatifs de la mise en place étendue de perrés à l'échelle des bassins hydrographiques.

1.0 INTRODUCTION

Rip-rap, the application of loose rock to streambanks, is commonly used in British Columbia as a streambank stabilization technique by private landowners, government and industry to prevent streambank erosion and to protect adjacent land-uses and infrastructure. For instance, as of 1995, approximately 40% (23 km) of the Coquihalla River and 38% (35 km) of the Coldwater River have been rip-rapped in order to protect adjacent land, land-uses and infrastructure. The prevalence of rip-rapped streambanks is due to their durability and effectiveness in preventing streambank erosion.

There is a range of potential positive and negative effects that rip-rap can have on fish and fish habitat. Positive effects can include reduced sediment inputs due to stabilization of eroding banks, increased cover due to the interstitial spaces along the edge of the rip-rapped bank, creation of deep pools due to scour along the armoured bank and improved fish passage along the armoured bank. Negative effects of rip-rapped streambanks can include a loss of riparian vegetation, resulting in a loss of nutrients and food sources, reduced future large woody debris (LWD) recruitment, and reduced shade. Other negative effects include decreased habitat diversity, a loss of in-channel habitat (rearing and spawning), a loss of gravel recruitment, and channelization which can result in downstream hydrological impacts, scouring of substrate (spawning gravels), loss of pool/riffle sequences, prevention of natural meanders, loss of floodplain high-water refuge habitat and access to off-channel rearing habitat.

Because of the range of positive and negative effects that rip-rap can have on fish and fish habitat, the Pacific Region of Fisheries and Oceans Canada (DFO) uses several different management approaches when dealing with rip-rap projects. Occasionally, a proposed rip-rap project will be considered a "harmful alteration, disruption, or destruction to fish habitat" (HADD). In these instances, DFO applies the Habitat Policy's (DFO 1986) guiding principle of no net loss (NNL) to the proposed rip-rap project so that a loss in the productive capacity of fish habitat will not occur. Under this principle, DFO can issue a Letter of Advice that indicates how the proponent is to relocate or redesign the proposed project to avoid the potential HADD and to fully mitigate any impacts the proposed development may have on fish habitat (DFO 1986, 1998). If, however, the HADD cannot be fully mitigated and it is determined that a proposed project will result in an unavoidable HADD and a loss of productive capacity of fish habitat, DFO can authorize a HADD under Section 35(2) of the *Fisheries Act* if it is in the public's best interest. To achieve NNL, DFO will require the proponent, through a legally binding authorization, to provide habitat compensation to offset the unavoidable losses of productive capacity as a result of the HADD.

In other instances, DFO treats rip-rap projects as restoration or creation projects. These rip-rap projects are designed to achieve the restoration and creation goals of the Habitat Policy by increasing the productive capacity of fish habitats to achieve an overall net gain (NG). This type of project can be implemented by DFO or by community and stewardship groups, but are not usually authorized under the Section 35(2) of the *Fisheries Act*.

The varied application of these habitat management approaches to rip-rap projects is evident in the fact that 65% of authorizations issued in British Columbia between 1994 to 1997 were for projects that involved the placement of rip-rap that constituted a HADD; in contrast, 20% of restoration projects implemented as part of the Pacific Region's Habitat

Restoration and Salmon Enhancement Program from 1998 to 2000 involved the incorporation of rip-rap with bio-engineered streambank protection strategies as a "fish-friendly" technique.

This study provides habitat management staff in DFO's Pacific Region with the most recent information on the range of potential effects of rip-rap on fish and fish habitat in order to improve the regional consistency of habitat management decisions regarding rip-rap installation. The report includes a comprehensive literature review of the effects of rip-rap on fish and fish habitat in the Pacific Northwest including an evaluation of over 20 years of environmental impact assessment monitoring data in British Columbia regarding the effects of rip-rap on fish and fish habitat. These two initiatives are presented in separate chapters followed by a discussion of the findings and recommendations for improving habitat management.

2.0
LITERATURE REVIEW ON THE EFFECTS OF RIP-RAP ON FISH AND FISH HABITAT WITH HABITAT MANAGEMENT IMPLICATIONS

by

D.A. Kochli[1] and J.I. Neville[1]

[1]G3 Consulting Ltd., 4508 Beedie Street, Burnaby, B.C. V5J 5L2

2.1 BACKGROUND

This literature review has been prepared to summarize current knowledge of the effects of rip-rap on freshwater fish and their habitats, particularly in British Columbia and the US Pacific Northwest. Often prescribed and implemented to stabilize stream banks and reduce erosion, rip-rap is a "hard engineering" solution with fish habitat management implications that are little understood. Rip-rap has been defined (USACE 1997) as a permanent, erosion-resistant ground cover of large, loose, angular stone with filter fabric or granular underlining. Heavy equipment is usually employed when placing rip-rap along stream banks, and given the weight of the material, it is not usually anchored in place.

2.2 PURPOSE AND APPLICATION OF RIP-RAP

The purposes of rip-rap are to protect streambank materials from the erosive forces of water, slow the velocity of concentrated runoff, and stabilize slopes with seepage problems or non-cohesive soils or both.

Rip-rap is usually applied at the soil and water interface when soil conditions, water turbulence and velocity, and expected vegetative cover suggest that the soil may erode under design flow conditions. Rip-rap may be applied at drain outlets, along stream channel banks or bottoms, along roadside ditches, in drop structures, at the toe of slopes, and as transition from concrete channels to vegetated channels (USACE 1997). In general, the application of rip-rap is an effective technique to stabilize stream channels and limit natural fluvial processes (Schmetterling *et al.* 2001).

During the past decade, increased recognition of the influences of rip-rap on stream hydrology has prompted a trend towards favoring softer bank stabilization methods, such as bioengineering techniques applied alone or in combination with rip-rap (e.g., live-stakes, wattles, or fascines). Principal disadvantages cited for rip-rap revetments are that they are unsuitable for steep slopes, may cause opposite banks to erode, and may damage aquatic or riparian habitat (MELP 2000). Furthermore, environmental impacts may be exacerbated if due attention is not given to such design considerations as material size and shape (Henderson 1986).

2.3 SCOPE OF THIS REVIEW

Rip-rap has widespread application. This review focused on rip-rap applied to banks of rivers and streams and its effects on fish and fish habitat. Other rip-rap applications (e.g., along a marine foreshore) were not considered.

In that context, this review attempted to answer the following overall questions:

1. What are the positive and negative effects of rip-rap on fish and fish habitat?
2. What mitigative measures are successful to offset negative effects?
3. What methods are appropriate to compensate for negative effects?
4. What are some alternatives to rip-rap?

2.4 METHODOLOGY

This project consisted of a review of recent literature including both primary and grey sources (government and consultant reports). We reviewed documents acquired through library and Internet searches and key papers cited in similar literature reviews (i.e., Craig and Zale 2001; and Schmetterling *et al.* 2001). Personal communications consisted of telephone interviews and E-mails to help identify and acquire certain research papers, and to discuss specific local issues. Cited contacts are listed in the reference section.

Much research on rip-rap focused largely on physical stability and appropriate engineering design. Of the limited literature available concerning implications of rip-rap for stream ecology, researchers tended to concentrate their attention on salmonids and game fishes, disregarding other aquatic life. In addition, studies were generally focused on specific sites or stream reaches and did not address cumulative effects at the watershed level.

2.5 RESULTS

Results of the literature review have been organized into sections that discuss positive effects of rip-rap (Section 2.1), negative effects of rip-rap (Section 2.2), mitigation of negative effects (Section 2.3), compensation measures (Section 2.4), and alternatives to rip-rap (Section 2.5). This chapter concludes with an overall summary discussion and recommendations for further research (Section 3.0).

2.5.1 Positive Effects of Rip-rap on Fish and Fish Habitat

Several positive effects of rip-rap on fish and fish habitat have been described. These include increased fish densities, higher fish abundance and higher fish diversity (Craig and Zale 2001). Positive effects were reported in large and small river systems, but were usually attributed to a somewhat degraded condition before rip-rap installation and the use of other mitigative measures used in combination with rip-rap (Craig and Zale 2001). Schmetterling *et al.* (2001) found that rip-rap can provide habitat for juvenile salmonids and bolster densities along reaches of streams that have been severely degraded. Whether effects were perceived as positive or negative depended on management objectives. A positive effect on one management objective (e.g., target species) may have negative implications for another objective. For example, rip-rap may provide cover for certain species but prevent natural channel meandering and future LWD and gravel recruitment, and therefore, decrease overall habitat quality (Craig and Zale 2001).

2.5.1.1 Stream Characteristics: Inconsistencies were found among studies investigating the effects of rip-rap on channel characteristics. Witten (1975) found that rip-rap revetments placed along Iowa streams to protect the bank had no apparent effect on stream morphology. No differences in velocity or maximum or minimum depth were evident. The author's description of the revetments was limited, though photos in the report suggested that the revetments investigated did not project into the channel and were similar to regular rip-rap. WDFW (2000) noted that armoured revetments created a

continuous, deepened thalweg and an uninterrupted high velocity pool along the treated bank. A deepened thalweg can have beneficial effects on fish densities in systems that have become de-stabilized, over-widened, and shallow (Binns 1994, Avery 1995, cited by Craig and Zale 2001).

2.5.1.2 Habitat Characteristics: Anecdotal observation (Binns 1994) suggested that, in Beaver Creek, Wyoming, during high flows (>50 ft^3/s or 1.4 m^3/s), crevices between rocks seemed to shelter many young trout; however, comparison between rip-rapped and natural banks was not attempted.

Streambanks with rip-rap tend to have less vegetation cover than natural streams (Schmetterling *et al.* 2001). In a study comparing 10 pairs of streams in Oregon and California, Hawkins *et al.* (1983) concluded that salmonids and sculpins were more abundant in streams without riparian shading than in shaded streams. Mean widths of streams studied ranged from 1.2 m to 5.9 m. Study authors cautioned against concluding that opening the canopy surrounding a stream would always result in a more desirable system, as the study focused only on small, open sections and the integrity of a natural stream is not solely determined by productivity of instream biota. Although environmental change may increase production, it may also cause shifts in community structure associated with loss of taxa typically found only in unmodified streams or alter the timing of life history events.

Craig and Zale (2001) observed that aquatic invertebrates flourish in rip-rap because it provides many interstices and high surface area and suggested that it may provide a superior food source for fish. These authors cautioned, however, that no studies have been conducted that directly show higher abundance of aquatic invertebrates in rip-rap benefit fish. Yet, in mostly channelized streams in Iowa, certain invertebrate species (notably mayflies and caddisflies) were found to be significantly more abundant on rock revetments than on unstructured control sites (Witten 1975). These invertebrates constituted a significant part of the diet of game fish found near the structures.

2.5.1.3 Multiple Species Assemblages: From a study of the upper Mississippi River, Madejczyk et al. (1998) concluded that "not only do certain species exhibit strong preferences for highly localized types of habitat, but fish assemblages within long stretches of channel border habitat are controlled to a large degree by dominant types of artificial habitat present." The main channel of the upper Mississippi River has undergone modification for more than 170 years. This study was conducted in a river section characterized by braided channels, forested islands, and a variety of backwater lakes and sloughs. Average discharge calculated was 703 m^3/s. Eighty-four species were collected, of which more than 40 percent were restricted to only one of three habitat types (wing dykes, woody snags or bare shore). Fish abundance and diversity differed little among habitat types. Significantly larger fish were present at locations with more complex structure than at sites with bare shore; fish biomass at individual wing dykes, woody snags and rip-rapped shorelines was found to be up to 30 times greater than that of bare shore sites.

2.5.1.4 Steelhead Trout: Knudsen and Dilley (1987) found that biomass of yearling and older steelhead increased dramatically in sections of two large streams (Lower Deschutes River and Decker Creek, Washington: mean discharge >4.9 m^3/s) after placement of rip-rap and attributed this finding to the pool-like characteristics in these sections and to the movement of juvenile steelhead downstream as they grew.

Over-wintering steelhead trout in the Coldwater and Nicola Rivers, BC, appeared to prefer rock crevices (provided by rip-rap or large substrate material) to off-channel and deep pool habitat (Swales *et al.* 1986).

2.5.1.5 Cutthroat Trout: Knudsen and Dilley (1987) observed that biomass of yearling and older cutthroat trout increased substantially in sections of two large streams (Lower Deschutes River and Decker Creek, Washington; mean discharge >4.9 m^3/s) after placement of rip-rap, while biomass decreased in the control sections. The authors concluded that the presence of rip-rap had improved cutthroat trout habitat.

In a long-term study of Huff Creek, Wyoming, Binns and Remmick (1994) examined several combinations of watershed management tools for enhancing populations of cutthroat trout *(Oncorhynchus clarki utah)* in stream reaches degraded by cattle overgrazing and resultant stream bank failures. Predicted bank-full flood flow of Huff Creek was 65 ft^3/s (~1.84 m^3/s). Several fish species were present, but only cutthroat populations were studied. Management tools included placement of in-stream structures and rip-rap and livestock control through enclosure and herding. The study concluded that "cutthroat trout were most abundant, and habitat was best, where both fencing and instream structures were used. Banks armoured with machine-placed rocks became stable; in contrast, natural healing was slow where rocks were not used." A combination of rip-rap placement and adequate riparian management seemed to be beneficial to fish populations, as high sediment inputs can have negative effects on fish and fish habitat (Birtwell 1999, Everest et al. 1986).

2.5.1.6 Brook Trout: In a long-term study of Beaver Creek, a second-order stream in the Black Hills National Forest, Wyoming, Binns (1994) found substantial increases in brook trout populations after 111 habitat improvement devices and 2,150 ft (~655 m) of rip-rap were installed. Beaver Creek had high flows greater than 50 ft^3/s (~1.4 m^3/s) and widespread bank erosion due to use of the streambed by cattle. Beaver Creek was chemically treated in 1970 to control non-game fish and then restocked with fingerling brook trout. Habitat structures were installed between 1973 and 1977. Fish populations were estimated before 1973 and for several years after until 1990. Three sites were studied: the treated site and untreated sections, both upstream and downstream. A variety of structures were installed at treated sites including rip-rap to stabilize eroding stream banks. Maintenance of structures was undertaken from 1974 to 1978, and again in 1985. The stream was 58% narrower in 1990 than in 1973, while pool volume was 323% greater and there were 200% more pools. By 1975, deep pools, pocket pools associated with rip-rap, and scour pools at deflectors and bank overhangs had provided a 338% increase in cover. In 1990, there were 183% more catchable brook trout than before habitat development (despite higher angling pressure) and the treated site contained 136% more trout than the upstream site. Although rip-rap was not the habitat unit preferred by brook tout compared to other habitat units (small pools, large pools and overhangs), the combination of rip-rap with other restoration techniques to stabilize banks appeared beneficial to brook trout populations in Beaver Creek. Binns (1994) also noted that instream devices would have been less effective had cattle grazing continued.

2.5.1.7 Brown Trout: Banks of Millevielle Creek, a small stream in Wisconsin (summer flow 7 ft^3/s to 10 ft^3/s [~0.2 m^3/s to 0.3 m^3/s]), was rip-rapped to counter effects of bank degradation caused by cattle grazing and row-crop farming in the riparian zone.

Following placement of rip-rap, the stream became deeper and density of brown trout increased significantly. The study showed that rip-rap can have beneficial effects on brown trout in degraded systems (Avery, 1995, cited by Craig and Zale, 2001).

2.1.3.6 Sockeye Salmon: Hinch and Rand (1998) observed that rip-rap placed to control erosion along the Nechako River, BC, generated small reverse flow fields (i.e., eddy vortices) along the shore that were used by adult sockeye salmon to facilitate their migration upstream. Migration through reaches with constrictions (e.g., large islands, gravel bars or large rock outcroppings) was found to require higher energy expenditure than that through reaches with parallel, straight banks. No attempt was made to compare migration within naturally straight banks and rip-rapped banks.

2.5.2 Negative Effects of Rip-rap on Fish and Fish Habitat

Three categories of negative effects of rip-rap have been described (Henderson, 1986; Craig and Zale, 2001):

- aquatic impacts (e.g., decreased fish densities, lower fish abundance, lower species diversity, favouring of introduced, exotic species, and loss of natural fluvial processes);
- terrestrial impacts; and
- aesthetic impacts.

Negative aquatic impacts have been reported in large and small systems, generally linked to systems that were pristine or minimally degraded before rip-rap was installed (Craig and Zale, 2001). Whether effects were perceived as positive or negative depended on management objectives. A negative effect on one management objective (e.g., target species) may in fact be a positive effect on another objective.

FISRWG (1998) provided a qualitative list of potential direct and indirect negative effects of several watershed activities. Activities listed included channelization and streambank armouring, and the effects of these activities included increased flow velocities, reduced capacity of a stream to assimilate nutrients or pesticides, and reduced biodiversity and biomass.

Schmetterling *et al.* (2001) highlighted indirect negative effects of rip-rapping, such as limited potential for large woody debris (LWD) and gravel recruitment, loss of undercut banks and overhanging vegetation, and displacement of native fishes by members of exotic species. These researchers also noted that, although rip-rap may provide habitat beneficial to invertebrates and juveniles of certain species, it does not provide the habitat required to support multiple age classes or species such as provided by natural banks that include LWD. As it is intended to stabilize channels, rip-rap limits natural fluvial processes and, therefore, has several deleterious effects on aquatic communities. Henderson (1986) deemed reduced habitat diversity to be the most serious adverse impact. Rip-rap projects may also impede future stream restoration efforts (Schmetterling *et al.*, 2001).

WDFW (2000) listed five types of impacts associated with bank protection projects:

- construction impacts;
- direct loss of habitat;

- channel response impacts;
- lost opportunity; and
- [illegible].

Construction impacts are generally short-term and can be mitigated by strategic timing and sequencing, protecting water quality, re-vegetating, and practicing erosion and sediment control. Long-term, adverse construction impacts may be incurred through removal of riparian vegetation, soil compaction, change of surface drainage, and the need to provide permanent access for maintenance equipment. Direct habitat loss may include loss of vegetation cover, loss of spawning beds, debris accumulation, reduced riparian function, and reduced habitat complexity and diversity. Channel response impacts can occur when a bank protection project influences the channel upstream and downstream of a site during future floods. Such effects are difficult to predict, but once they occur, typically persist and propagate, leading to the need for additional bank protection. Lost opportunities can include constraints to channel migration and therefore prevent creation of off-channel habitat, side channels, flood channels, floodplain connectivity, and recruitment of gravel and debris. Bank protection may lead to a perception that properties adjacent to the channel are safe, and subsequently result in increased floodplain development.

2.5.2.1 Stream Characteristics: Bank stabilization structures alter not only the banks they are designed to protect, but also the morphology and physical structure of a river. In small rivers, channelized sections have been reported to be uniformly shallow and homogeneous (Elser 1968; cited by Craig and Zale 2001). In larger rivers, bank stabilization has been reported to reduce the number of islands and braided reaches, decrease the channel width and deepen the riverbed (several authors cited by Craig and Zale 2001). Bank stabilization structures can also restrain the natural lateral migration of a river channel, thereby reducing the potential for LWD input, substrate deposition, pool-riffle development, and sidechannel formation. It has also been shown that bank stabilization and channelization also reduced the frequency of inundation of side channels (Stern and Stern 1980; cited by Craig and Zale 2001) and backwaters. These impacts would be expected to decrease habitat quality for fish and limit their abundance and production (Craig and Zale 2001 and Schmetterling et al. 2001).

2.5.2.2 Habitat Characteristics: Several authors generally recognized that rip-rap reduced habitat complexity and diversity, important to survival, growth, migration, and reproduction of salmonids. Negative effects of rip-rapped streambanks can include a loss of riparian vegetation, resulting in a loss of nutrients and food sources, decreased future LWD recruitment, and reduced shade, and a decrease in habitat diversity. Placement of rip-rap without revegetating a site reduces the amount of riparian vegetation on a stream, and rip-rapped banks may remain barren for an extended time compared to natural banks (several authors cited by Schmetterling *et al.* 2001).

Other negative effects include a loss of in-channel habitat (rearing and spawning) and a loss of gravel recruitment. Natural channel and bank erosion provides gravel recruitment needed for spawning. Rip-rap may reduce gravel supply. Changes to the bank can also cause the thalweg to deepen, scouring gravel and creating uniform channel beds devoid of spawning habitat.

2.5.2.3 Multiple Species: Peters *et al.* (1998) conducted snorkel surveys at 67 sites along several rivers in western Washington to study fish densities near banks stabilized through various means. Mean discharge ranged from approximately 7 m^3/s to 80 m^3/s. Five stabilization methods were distinguished: rip-rap, rip-rap with LWD, rock deflectors, woody debris (LWD-revetment) and rock deflectors with LWD. For each site a naturally stable control site with similar channel characteristics was chosen. Fish species studied included salmonids, redsided shiner, dace and sculpins. Fish densities were linked to stabilization method and such habitat variables as riparian cover, vegetation overhang, and area and number of LWD pieces. It was concluded that sites treated with rip-rap consistently had lower fish densities than control sites during all surveys, and that sites stabilized with LWD were the only project types that consistently had greater fish densities than control areas. The researchers recommended that LWD cover be used whenever possible and that LWD incorporated into rip-rap and rock deflectors be larger and more complex than that previously used. The authors also noted that different fish species exhibit different responses to bank stabilization and those responses change seasonally, emphasizing the need to know the species in the system in question.

2.5.2.4 Coho & Chinook Salmon: Placement of rip-rap in large streams (Western Washington; mean discharge >4.9 m^3/s) appeared to adversely affect juvenile coho salmon densities (Knudsen and Dilley 1987). Juvenile coho over-wintering in the Coldwater River were significantly more abundant in natural main-channel pools than in main-channel sites altered by rip-rap (Swales *et al.* 1986). In the same study, over-wintering juvenile chinook salmon were also significantly more abundant in natural main-channel pools than rip-rap sites.

2.2.3.3 Age-Zero Steelhead & Cutthroat Trout: Biomass of young-of-the-year (age-0) steelhead and cutthroat trout decreased slightly after rip-rap was placed along large streams in Western Washington (mean discharge >4.9 m^3/s) (Knudsen and Dilley 1987).

2.5.3 Mitigation of the Negative Effects

Mitigation measures have sought to alter rip-rap design to reduce anticipated negative effects on fish and fish habitat. No universal formula was evident for calculating mitigation required for any one project (WDFW 2000). According to MELP (2000), mitigative measures are site-specific and must address the particular species and life stages that use the habitat affected by bank protection works. A number of features are thought to contribute to improving rip-rap revetments, including: scalloping the low water shoreline; placing large rock at the toe; using rocks larger than required for hydraulic stability; adding spurs; incorporating planting baskets or eco-pockets; or building stepped benches for riparian vegetation (MELP 2000). These measures are further discussed below. Lister *et al.* (1995) concluded that, rip-rap design for habitat enhancement must be carefully considered and be based on biological and hydraulic requirements, that the pattern of habitat utilization by fish should be known, and that no one design prescription will be appropriate for all situations.

2.5.3.1 Size of Rip-rap: A study of the Thompson and Coldwater rivers, BC, found that banks rip-rapped with material of mean diameter greater than 30 cm

supported higher chinook salmon, coho salmon and steelhead trout densities than banks rip-rapped with smaller material (Lister *et al.*1995). Craig and Zale (2001) cited other studies that indicated positive and negative effects of rip-rap, depending on rock size and fish species studied. For example in the Skagit river (Washington), juvenile rainbow trout were found to be more abundant along banks with boulder-size rip-rap (~25.6 cm) than along natural banks, but the reverse was true for banks with cobble-size rip-rap (6.4 cm to 25.6 cm; Beamer and Henderson, 1998, cited by Craig and Zale 2001). Those studies suggest that increased rock sizes could be beneficial for several fish species. Rock size and revetment thickness could often be increased without much additional cost (MELP 2000). Froehlich and Benson (1996) published a simple method for sizing rip-rap and estimating reliability of a protected slope.

2.5.3.2 Habitat Complexing at Toe of Slope: Large boulders installed along the toe of armoured banks were noted to dissipate energy and greatly improve habitat capacity (Finnigan and Slaney 1999) by increasing flow diversity and creating low velocities, eddies and shear zones (MELP 2000). Boulders of 1.0 m to 1.5 m diameter appeared to increase rearing densities of all salmonids except sub-yearling steelhead trout along the Coldwater River in BC (Lister *et al.* 1995).

2.5.3.3 Spurs or Groins: In a study of the Sacramento River, California, the USFWS (1992; cited by Craig and Zale 2001) determined that fish groins running perpendicular to a channel, from the toe to the top of the bank, were used by juvenile salmonids more than unimproved rip-rap.

2.5.3.4 Angularity of Rip-rap: In Beaver Creek, Wyoming (high flows >50 ft^3/s or >1.4 m^3/s), Binns (1994) determined that angular rocks larger than 2 feet (~0.6 m) in diameter best stabilized eroding stream banks. Floods washed away smaller rocks and often shifted round or flat rocks out of position. Angular rocks contributed to rip-rap stability, thereby preventing further impact on the watercourse. Angular rocks provide crevice habitat important for rearing fish.

2.5.3.5 Scalloping of Rip-rap: By scalloping rip-rap, small embayments are created along the waterline by increasing and decreasing the slope of the revetment. Embayments typically create approximately 10 m^2 of low-velocity rearing habitat with eddies and shear zones. This approach was found to be most suitable along steep, gravel-bed streams that lacked areas of low velocity (MELP, 2000).

2.5.3.6 Slope of Rip-rap: Apart from the technique of changing slope to scallop rip-rap (described above), no literature was found linking rip-rap slope to effects on fish and fish habitat. Lowering the angle of rip-rap to a maximum of 3H:1V has been recommended, however, for recreational purposes (including angling; MELP 2000), a practice that could have indirect impacts on fish populations.

2.5.3.7 Terracing & Joint Planting with Riparian Vegetation: Though several techniques have been developed for offsetting loss of riparian vegetation when rip-rap is installed, no research appears to have provided technical details for measures that would completely mitigate negative impacts (e.g., most suitable species or extent and location of planting).

MELP (2000) described terracing, a technique by which a bench is formed part way up a revetment slope and planted with riparian vegetation. Integrated planting baskets or

"ecopockets", consisting of soil retaining containers (e.g., gabions), were placed within rip-rap revetments and planted with riparian vegetation after construction (MELP 2000). Donat (1995) described the technique of planting vegetation amongst rip-rap, and its advantages and disadvantages, but did not relate this practice to fisheries issues. In certain situations, additional vegetation on or near flood protection works appeared not to compromise structural integrity or public safety. Controlled vegetation clumps are acceptable in rip-rap bank protection under the *Environmental Guidelines for Vegetation Management on Flood Protection Works in BC* (MELP AND DFO 1999).

2.5.3.8 Incorporation of LWD: Peters *et al.* (1998) studied fish densities in Western Washington near riverbanks stabilized by various methods (see section 2.3.4; Fish Species & Life Phases). It was recommended that LWD cover be used whenever possible and that LWD incorporated into rip-rap be as large and complex as possible.

2.5.4 Compensation Measures

Compensatory habitat may be provided elsewhere when adverse project impacts cannot otherwise be fully mitigated on-site. Anticipated loss of habitat due to rip-rap installation has been described above, and includes loss of undercut banks, loss of riparian function, and loss of spawning gravel. Compensatory measures should be site-specific and address the particular species and life stages that use the habitat affected by bank protection works. Little literature was found dealing directly with compensation issues.

2.5.4.1 Gravel Placement: Artificially supplementing a channel with spawning gravel could compensate for lowered gravel recruitment or gravel scouring caused by bank protection works (WDFW 2000). Careful design is required to ensure newly placed substrate will not be flushed out (Slaney and Zaldokas 1997), and gravel may need to be regularly replaced (Sheng *et al.* 1998).

2.5.4.2 Off-Channel and Side-Channel Development: When a project eliminates off-channel or side channel habitat, or reduces the opportunity for such habitat to form, compensatory habitat should be created, restored, or enhanced. Bates (2001, pers. com.) recommended creation and re-activation of off- and side-channels to compensate for lost opportunities. Lost-opportunity impacts could be avoided by selecting a deformable bank protection technique (WDFW 2000). Off-channel and side-channel habitat has been successfully created for target species in several cases (e.g., Lister 1997, Swales and Levings 1989).

2.5.5 Alternatives to Rip-rap

Depending on the site, several bank stabilization techniques could be considered as alternatives to rip-rap. Stability and longevity of these techniques have been little documented and are not well known however, and such measures are recommended for streams with low to moderate velocities only (MELP 2000). Non-rip-rap alternate designs appeared appropriate where the main management objective was to restore fish habitat rather than prevent bank erosion and damage to infrastructure (MELP 2000).

Doyle (1992) discussed stability of four alternative designs on seven sites in BC (Coldwater, Salmon, and Perry Rivers) and concluded that, on steep gravel-bed rivers,

gravel dykes do not endure; tree revetments require constant maintenance; toe aprons are not as reliable as toe trenches; and well-placed, large semi-round rock performs well. Doyle (1992) did not determine when a less expensive or more environmentally acceptable alternative would offer a good chance of longevity.

Benefits of bioengineering methods compared to traditional engineering techniques included technical, ecological, economic and aesthetic advantages and make these techniques worthwhile for consideration (Donat, 1995).

2.5.5.1 Debris Groins: Finnigan and Slaney (1999) reported how debris groins (fabricated logjams) were applied as alternatives to rip-rap on the West Kettle River, BC. Debris groins resemble natural templates more closely and seem to be "more effective in shifting the thalweg sufficiently to allow sediment deposition and re-vegetation". Natural lateral log jams (templates for debris groins) tended to be "associated with lateral scour pools, providing prime fish-holding and rearing sites in all sizes of streams, but especially in medium or larger streams". The study did not define stream sizes, nor was any methodical study undertaken to assess effects on fish habitat and fish populations.

2.5.5.2 Rootwad & Boulder Revetments: Installation of logs with intact rootwads was found to re-establish natural roles of large woody debris in streams by creating a dynamic near-bank environment that traps organic material and provides colonization substrates for invertebrates and refuge habitats for fish (FISRWG, 1998). As logs eventually rot, this technique stabilizes the bank only temporarily. Therefore, timely establishment of functional riparian vegetation is crucial (FISRWG, 1998).

2.5.5.3 Current Deflectors: Current deflection structures have been reported to be the primary alternative to rip-rap (Craig and Zale 2001) and to provide habitat benefits, as they require less bank preparation and create low-velocity habitat beneficial to aquatic animals (Henderson, 1986). Though fish habitat value of reaches that lack diversity could likely be enhanced by adding deflectors, it was cautioned that deflectors may lead to riverbed degradation and de-watering of side-channels (Craig and Zale 2001).

2.5.5.4 Spurs and Groins: Spurs may be used instead of continuous revetment to prevent erosion. These structures may not provide complete protection. Spurs and groins could have potential adverse effects on stream hydraulics, navigation, ice passage, and fish passage and should receive careful consideration in design (MELP, 2000). Donat (1995) described several groin designs and their hydraulic implications, but did not specifically address fisheries concerns. USEPA (2001) described groins with willow planted in the depositional zone but undertook no methodical studies of their effects on fish and fish habitat.

2.5.5 Other Alternative Techniques: Though several alternative techniques have been described in literature, effects of most on fish and fish habitat were not methodically studied. Generally, however, measures were aimed at improving riparian habitat or aesthetics, or were deformable in design (WDFW 2000). A brief summary of some interesting designs is provided below.

- Donat (1995) and FISRWG (1998) described several alternative bioengineering techniques including vegetated gabions, brush mattresses, live stakes, live

fascines, and vegetated geogrids. In addition, FISRWG (1998) described and provided further references for such techniques as coconut fibre rolls, dormant post planting, stone toe protection, and joint plantings.

- Dorava (1999) described bio-logs as a successful technique to reduce wake-induced riverbank erosion along the Kenai River, Alaska.

- Live cribwalls are described as hollow, box-like interlocking arrangements of logs filled with alternate layers of soil and live cuttings that root and gradually take over structural functions of wood members (Donat, 1995; FISRWG, 1998). MELP (2000) proposed making cribwalls more "habitat-friendly" by modifying them to include LWD pieces with rootwads projecting into the stream.

- Adams (2002) provides a review of environmental design considerations in the Shoreline Structures Environmental Design manual (http://www.stewardshipcentre.bc.ca/sc_bc/main/index.asp?sProv=bc#).

2.6 DISCUSSION AND RECOMMENDATIONS

Craig and Zale (2001) attributed conflicting findings of rip-rap studies to diverse pre-existing conditions: rip-rap seemed beneficial in systems already somewhat degraded and deleterious when applied in relatively pristine systems, no matter of their size. In addition, physical descriptions of rip-rap in many studies tended to be incomplete or vague, making it difficult to discern distinctions. Craig and Zale (2001) also noted that "differing effects of rip-rap in different studies may also have been an artifact of when those studies were conducted and which life stages or species were focused on." Peters *et al.* (1998) emphasized the need to know the species in the system in question, because "different fish species show different responses to bank stabilization and those responses change seasonally." In some studies, certain fish species or life stages benefited from rip-rap while others did not. Limitation of natural fluvial processes was of concern for many authors, but has not been studied methodically (e.g. Schmetterling *et al.*, 2001). "Integrated streambank protection requires a change in the traditional approach; bank protection measures should be selected to address site- and reach-based conditions and to avoid habitat impacts rather than automatically applying traditional methods such as rip-rap" (Carmer et al., 2000).

2.6.1 Management Objectives

In systems where excessive erosion had occurred, installing rip-rap reduced erosion and stabilized banks, thus providing a benefit to fish populations (Craig and Zale 2001, Schmetterling *et al.* 2001). It was also beneficial in large, channelized rivers as a measure for increasing habitat complexity (Madejczyk *et al.* 1998).

Generally, however, studies concluded that rip-rap was limiting natural fluvial processes and therefore was adversely affecting many aspects of fish habitat (e.g. Bates, 2001, pers. com.). Because different species react differently, it is important to know detailed management objectives for streams and watersheds potentially affected by rip-rap. If consequences are not known, mitigative and compensatory measures cannot be designed properly. Given that a lack of knowledge of cumulative effects in specific

watersheds is common, it is recommended that a precautionary approach be applied and that natural fluvial processes be encouraged whenever possible. For each stream, an appropriate migration corridor should be designated and land-use planning undertaken accordingly. Stream banks close to the boundaries of this corridor would be protected by permanent structures (e.g., rip-rap). Banks within the allowed corridor would be protected with deformable structures, allowing for natural processes (WDFW, 2000).

If management objectives dictate a permanent structure, applying mitigative measures as described above would enhance rip-rap. Permanent rip-rap alternatives should be considered as well (e.g., current deflectors, spurs and groins).

2.6.2 Alternatives to Rip-rap

Planners and designers of streambank protection projects should consider causes of bank failure, habitat and risk (Carmer *et al.*, 2000), as well as location within the natural stream migration corridor (WDFW, 2000). Depending on these factors, there may be better alternatives to rip-rap. Schmetterling *et al.* (2001) concluded that use of rip-rap should be discontinued in favour of natural materials to allow fluvial processes (e.g., lateral stream bank erosion) to occur. These authors also predicted a continued struggle for resource managers trying to maintain natural fluvial processes while protecting public infrastructure and private property from those same processes.

2.6.3 Research Recommendations

Schmetterling *et al.* (2001) concluded that the practice of rip-rapping banks goes against current practices and philosophies of stream re-naturalization and impedes future restoration work. It is therefore recommended that a detailed review of policies currently applied in other North American jurisdictions be undertaken. Analysis would focus on management objectives, political tools (e.g. legislation, best management practices, and guidelines), and success of policies.

On a scientific level, no studies were identified that comprehensively addressed long-term effects of bank stabilization over large spatial scales (Craig and Zale, 2001) and cumulative effects of rip-rap at the watershed level appeared to have been overlooked by researchers (Schmetterling *et al.*, 2001). Future research should also determine the effects of rip-rapped banks on salmonid densities, and the effectiveness of soft techniques such as bioengineering for stabilizing streambanks.

2.7 REFERENCES

2.7.1 Personal Communications

Bates, K. 2001. Chief Environmental Engineer. Washington State. Telephone conversation,May 24, 2001.

Peters, R.J. 2001. US Fish and Wildlife Service, North Pacific Coast Ecoregion, Western Washington Office, Aquatic Resources Division, Lacey, Washington. Telephone conversation, April 27, 2001.

2.7.2 Electronic Sources

USEPA. 2001. US Environmental Protection Agency. Rock Barbs Enhance Fish Habitat and Water Quality in Oregon's Tillamook Bay Watershed. http://www.epa.gov/OWO/estuaries/coaslines/spring98/rockbarb.html (accessed 03 May, 2001).

WDFW. 2000. Washington Department of Fish and Wildlife. Integrated Streambank Protection Guidelines. http://www.wa.gov/wdfw/hab/salguide/salguide.htm. (accessed 08 May, 2001).

2.7.3 Literature Reviewed

Avery, E.L. 1995. Effects of streambank riprapping on physical features and brown trout standing stocks in Millville Creek. Research Report 167. Wisconsin Dept. of Natural Resources.

Beamer, E.M., and R.A. Henderson. 1998. Juvenile salmonid use of natural and hydromodified stream bank habitat in the mainstem Skagit River, northwest Washington. Skagit System Cooperative. LaConner, Washington.

Binns, A.N. 1986. Stabilizing eroding stream banks in Wyoming - A guide to controlling bank erosion in streams. Wyoming Game and Fish Department. Cheyenne, Wyoming; 42 pp.

Binns, A.N. 1994. Long-term responses of trout and macrohabitats to habitat management in Wyoming headwater streams. N. Am. J. Fish. Manage. 14(4):87-98.

Binns, A.N., and R. Remmick. 1994. Response of Bonneville cutthroat trout and their habitat to drainage-wide habitat management at Huff Creek, Wyoming. N. Am. J. Fish. Manage. 14(4):669-680.

Bjornn, T.C., and D.W. Reiser. 1991. Habitat requirements of salmonids in streams. *In:* W.R. Meehan (ed.). Influences of forest and rangeland management on salmonid fishes and their habitats. USDA and USFS. Bethesda, MD; pp. 83-138.

Carmer, M., K. Bates, and D.E. Miller. 2000. Integrated streambank protection guidelines. Washington Department of Fish and Wildlife and Inter-Fluve Inc. Olympia, WA; 5 pp.

Craig, A.J., and A.V. Zale. 2001. Effects of bank stabilization structures on fish and their habitat - A literature review. US Geological Survey, Washington Department of Ecology, and Montana State University. Bozeman, MT; 29 pp.

Culp, J.M., and R.W. Davies. 1983. An assessment of the effects of streambank clearcutting on macroinvertebrate communities in a managed watershed. Can. Tech. Rep. Fish. and Aquat. Sci. 1208. Nanaimo, BC; 87 pp.

Donat, M. 1995. Bioengineering techniques for streambank restoration; A review of central European practices. Watershed Restoration Project Report 2. MELP. Vancouver; 86 pp.

Dorava. 1999. Effectiveness of streambank -stabilization techniques along the Kenai River, Alaska. Water-Resources Investigations Report 99-4156. USGS. Anchorage, Alaska; 21 pp.

Doyle, P.F. 1992. Performance of alternative methods of bank protection. Can. J. Civ. Eng., 19:1049-1061.

Elser, A.A. 1968. Fish populations of a trout stream in relation to major habitat zones and channel alterations. Trans. Am. Fish. Soc. 97:389-397.

Finnigan, R., and P.A. Slaney. 1999. Channel rehabilitation: Debris groins as a bank stabilization option. Streamline 4(2):17-19.

FISRWG. 1998. Stream corridor restoration: Principles, processes, and practices USDA. Washington, DC; 548 pp.

Froehlich, D.C., and C.A. Benson. 1996. Sizing dumped rock riprap. J. Hydraulic Eng. 122(7):389-396.

Hawkins, C.P., M.L. Murphy, N.H. Anderson, and M.A. Wilzbach. 1983. Density of fish and salamanders in relation to riparian canopy and physical habitat in streams of the northwestern United States. Can. J. Fish. Aquat. Sci. 40(8):1173-1185.

Henderson, J.E. 1986. Environmental designs for streambank protection projects. Water Res. Bull. 22(4):549-558.

Hinch, S.G., and P.S. Rand. 1998. Swim speeds and energy use of upriver-migrating sockeye salmon (*Oncorhynchus nerka*): Role of local environment and fish characteristics. Can. J. Fish. Aquat. Sci. 55:1821-1831.

Knudsen, E.E., and S.J. Dilley. 1987. Effects of riprap bank reinforcement on juvenile salmonids in four western Washington streams. N. Am. J. Fish. Manage. 7:351-356.

Lister, D.B. 1997. No net loss and the Coquihalla Highway: Along the Coldwater River. *In:* No net loss of habitat: assessing achievement. Kwantlen University College and DFO. Richmond, BC; pp. 17-19.

Lister, D.B., R.J. Beniston, R. Kellerhals, and M. Miles. 1995. Rock size affects juvenile salmonid use of streambank riprap. *In:* C. R. Thorne, S.R. Abt, F.B.J. Barends, S.T. Maynord, and K. W. Pilarczyk (eds.). River, coastal and shoreline protection: Erosion control using riprap and armourstone. John Wiley & Sons Ltd. pp. 621-632.

Madejczyk, J.C., N.D. Mundahl, and R.M. Lehtinen. 1998. Fish assemblages of natural and artificial habitats within the channel border of the upper Mississippi River. Am. Midland Nat. 139(2):1-296.

MELP and DFO. 1999. Environmental guidelines for vegetation management on flood protection works to protect public safety and the environment. Surrey, BC; 17 pp.

MELP. 2000. Riprap design and construction guide. Public Safety Section, Water Management Branch. Victoria, BC; 65 pp.

Peters, R.J., B.R. Missildines, and D.L. Low. 1998. Seasonal fish densities near river banks stabilized with varoius stabilization methods; first year report of the flood technical assistance project. USFWS. Lacey, WA; 34 pp.

Schmetterling, D.A., C.G. Clancy, and T.M. Brandt. 2001. Effects of riprap bank reinforcement on stream salmonids in the western United States. Fisheries 26(7):6-11.

Sheng, M., S. Anderson, and R. Norgan. 1998. Campbell River spawning-gravel placement project. Streamline 3(1):1-6.

Slaney, P.A., and D. Zaldokas. 1997. Fish habitat rehabilitation procedures. Watershed Restoration Technical Circular 9. MELP. Vancouver, BC; 309 pp.

Stern, D.H., and M.S. Stern. 1980. Effects of bank stabilization on the physical and chemical characteristics of streams and small rivers: a synthesis. USFS and Missouri Institute of River Studies. U.S. Fish and Wildlife Service Report FWS/OBS-80/11.

Swales, S., and C.D. Levings. 1989. Role of off-channel ponds in the life cycle of coho salmon (Oncorhynchus kisutch) and other juvenile salmonids in the Coldwater River, British Columbia. Can. J. Fish. Aquat. Sci. 46:232-242.

Swales, S., R.B. Lauzier, and C.D. Levings. 1986. Winter habitat preferences of juvenile salmonids in two interior rivers in British Columbia. Can. J. Zool. 64:1506-1514.

USACE. 1997. Engineering and design-handbook for the preparation of storm water pollution prevention plans for construction activities. EP 1110-1-16.

USFWS. 1992. Juvenile salmon study, Butte Basin Reach, Sacramento River Bank Protection Project. Prepared for USACE. Sacramento, CA.

Witten, A.L., Jr. 1975. A study of the impact of selected bank stabilization structures on game fish and associated organisms. Master Thesis, Iowa State University, Department of Animal Ecology. Ames Iowa; 63 pp.

Acronyms used in references:

DFO: Canadian Department of Fisheries and Oceans (Fisheries and Oceans Canada)
FISRWG: Federal Interagency Stream Restoration Working Group
MELP: British Columbia Ministry of Environment, Lands and Parks
USACE: United States Army Corps of Engineers
USDA: United States Department of Agriculture
USFS: United States Forest Service
USFWS: United States Fish and Wildlife Service

3.0
FIELD EVALUATION OF RIP-RAP EFFECTS ON JUVENILE SALMONID HABITAT

By

D.B. Lister[1]

[1] D.B. Lister & Associates Ltd., P.O. Box 2139, Sardis Stn Main, Chilliwack, B.C. V2R 1A5

3.1 INTRODUCTION

One of the principal pieces of information missing from the scientific literature relates to the effect of rip-rap on juvenile salmonid densities (Schmetterling et al. 2001). To help address this gap in the literature, over seven years of environmental impact assessment data relating to rip-rap developments on four rivers in British Columbia were compiled. Data were analyzed to determine when rip-rap was beneficial to juvenile salmonids and under what conditions it was harmful.

3.2 FIELD STUDY AREAS

Studies relating to effects of rip-rap revetments on juvenile salmonid habitat were carried out in four areas of the Fraser River Basin: North Thompson River; Thompson River; Coldwater River; and lower Fraser River (Figure 1). Study area locations, characteristics and fish populations are described below.

North Thompson River studies were conducted in 2 separate mainstem areas: (1) an upper river study area including 24 km of river mainstem between Lempriere and Thunder River, situated approximately 15 km north of the town of Blue River; and (2) a lower river study area along 140 km of river mainstem from Avola downstream to McLure. River channel width is generally 50 m – 200 m, but can be up to 300 m in reaches of the lower river study area. The North Thompson is a large, glacially-fed river with mean annual discharge in the lower study area ranging from 149 m^3/s near the upstream limit (Birch Island gauge) to 428 m^3/s at the downstream limit (McLure gauge; WSC 1989a). Chinook (*Oncorhynchus tshawytscha*) and coho (*O. kisutch*) salmon and mountain whitefish (*Prosopium williamsoni*) are the most abundant salmonids in the study area (Whelen and Lister 1985a, 1985b). Small numbers of juvenile sockeye salmon (*O. nerka*), rainbow trout (*O. mykiss*), bull trout (*Salvelinus confluentus*), redside shiner (*Richardsonius balteatus*), sculpins (*Cottus sp.*), suckers (*Catostomus sp.*) and lamprey (*Lampetra sp.*) are also present. Coho salmon were the only anadromous species captured in the upper river when the present study was conducted in 1984 and 1985 (Whelen and Lister 1985a).

The Thompson River study area included river mainstem sites in the 100 km section between Kamloops Lake and Spences Bridge. In this section the wetted channel is 100 m - 200 m wide and carries an estimated mean annual discharge of 775 m^3/s (Spences Bridge gauge; WSC 1989a). Chinook salmon and rainbow-steelhead trout (*O. mykiss*) are the principal juvenile salmonids in the mainstem (Beniston et al. 1985). Mountain whitefish are common and small numbers of juvenile coho and sockeye salmon are also present. Fish sampling also revealed the presence of suckers, sculpins, redside shiners, lamprey and longnose dace (*Rhinichthys cataractae*) (Beniston et al. 1985). The *O. mykiss* population may have included both resident rainbow trout and anadromous steelhead which could not be readily separated in the field. For simplicity, these two life history types are identified as rainbow trout.

Coldwater River is a tributary of the Nicola River which flows into the Thompson River at Spences Bridge. The stream has a 90 km mainstem length between headwaters on the east slope of the Cascade Mountains and its confluence with the Nicola River at Merritt. The work reported here was conducted in a 39 km river section between Henning Bridge, the first highway bridge east of Coquihalla Lakes, and Kingsvale which is

situated approximately 30 km upstream of the mouth. Mean annual discharge at Brookmere, near Kingsvale, is estimated at 6.7 m^3/s (WSC 1989a). Wetted stream width is generally 10 m – 15 m under low flow conditions in late summer. Salmonid fish, principally chinook salmon, coho salmon and steelhead, dominate the rearing and over-wintering juvenile fish community in the Coldwater mainstem (Beniston et al. 1988). Bull trout and mountain whitefish are also present.

The lower Fraser River study was conducted along the south bank of the mainstem in 2 sections: (1) a 26 km reach between Hope and Agassiz; and (2) a 25 km section from the Sumas River confluence to 10 km downstream of Mission. In this area, flood plain width ranges from approximately 300 m to 3000 m; wetted channel width varies greatly with river discharge and location. Mean annual discharge of the Fraser River at Hope averages 2720 m^3/s (WSC 1989a). Water quality is characterized by seasonally high suspended solids levels, up to 1460 mg/L, during snowmelt freshet in May-July (WSC 1980). Juvenile chinook salmon are the predominant salmonid fish species rearing in the Fraser River mainstem (Beniston et al. 1986). Coho juveniles are common only near the Sumas River confluence, and relatively low numbers of juvenile sockeye salmon, chum salmon (*O. keta*), rainbow trout, coastal cutthroat trout (*O. clarki clarki*) and mountain whitefish are also present. Non-salmonid fish species in the study area include three-spine stickleback (*Gasterosteus aculeatus*), sculpins, suckers, redside shiner, longnose dace, leopard dace (*Rhinichthys falcatus*), lamprey, northern pikeminnow (*Ptychocheilus oregonensis*) and peamouth chub (*Mylocheilus caurinus*) (Beniston et al. 1986).

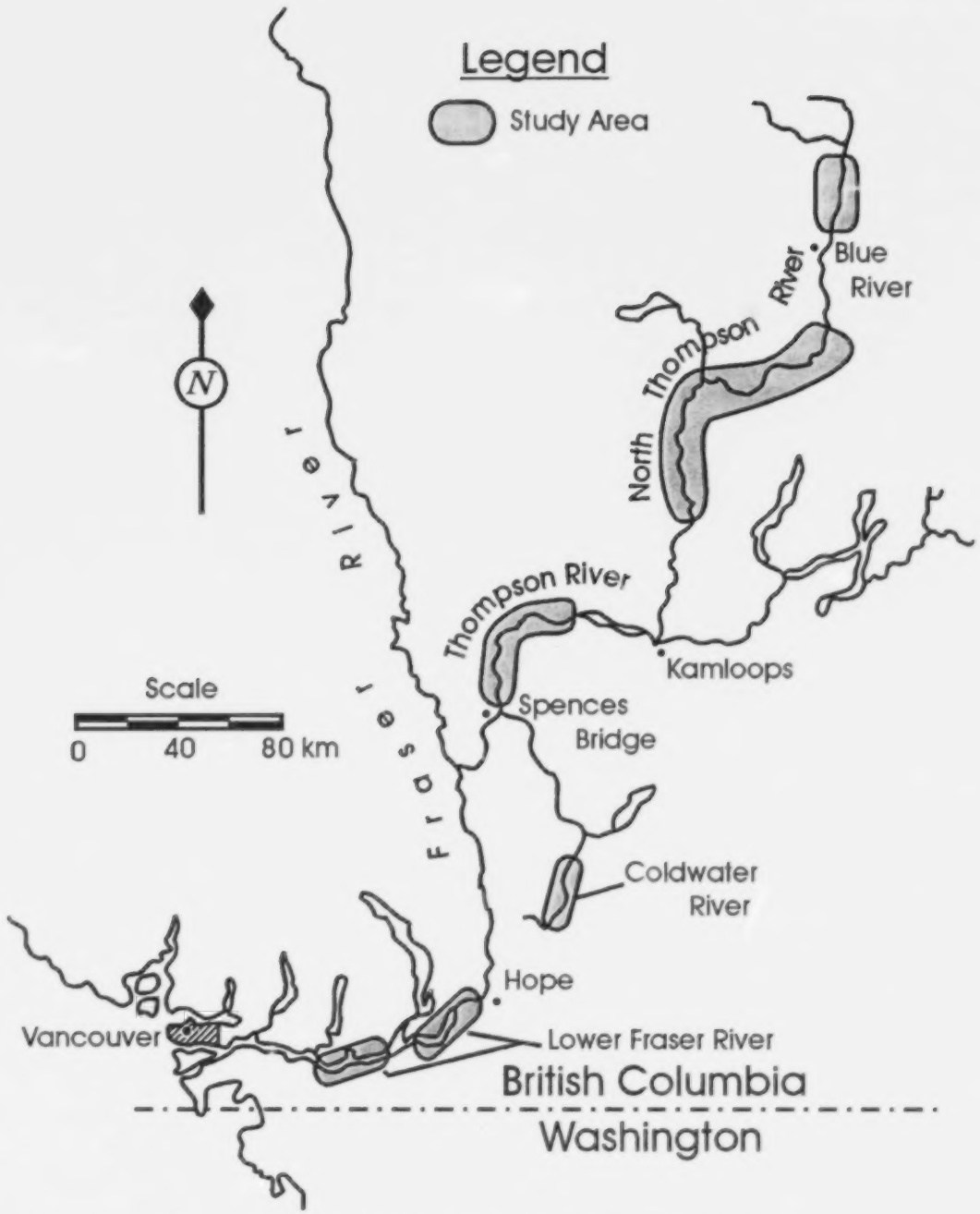


Figure 1. Study area locations in the lower Fraser River Basin.

3.3 CASE STUDIES

This study involved juvenile fish sampling and habitat characterization in the 4 study areas described in the preceding section. A basic assumption of the study is that the level of habitat use by salmonid juveniles indicates relative suitability of the habitat for rearing or over-wintering. In the 3 large river study areas, the North Thompson, Thompson and lower Fraser rivers, the study approach was to compare fish use of steep river banks with or without rip-rap protection. Fish sampling sites at rip-rap banks were all situated along railway embankments. Sampling sites without rip-rap protection, located at or near railway embankments, were steep banks where rip-rap had not been added and, in many cases, where natural revegetation had occurred. While non-rip-rap banks are generally identified here as *natural* banks, it should be recognized that these sites may have been altered at some time by railway construction or maintenance activities.

The study design employed at Coldwater River differed from that used in large river studies. Fish use of 6 stream sections altered by highway construction, i.e., stream diversion or encroachment, was compared to fish use of adjacent unaltered stream sections selected to represent physical conditions at each altered site prior to highway construction.

In the North Thompson River and lower Fraser River studies, the effects of rip-rap on juvenile salmonid habitat were assessed by comparing fish utilization of unpaired rip-rap and natural bank sites. Though an attempt was made to consider effects of other physical factors, e.g. water velocity or depth, data analysis could not entirely account for the non-random distribution of rip-rap treatments. Along railway embankments, such as those on the North Thompson and Fraser rivers, rip-rap has been placed at sites where bank erosion had occurred. Such sites are relatively exposed to strong flows and subject to streambed scour. Rip-rap study sites therefore tended to have higher velocities and depths than the natural bank sites to which they were compared. Given the extreme variation in fish abundance data, it was not possible to completely isolate effects of physical circumstances, such as water depth and velocity, from effects of bank type (rip-rap or natural). Design of studies at the Thompson and Coldwater rivers, based on paired sampling sites with similar water depth or velocity, was intended to overcome this problem.

3.4 NORTH THOMPSON RIVER

3.4.1 Methods

Fish sampling was conducted in late summer (August 21 – September 23, 1984) and late winter (March 19-30, 1985). Daily discharge of the North Thompson River at Birch Island ranged from 94 m^3/s to 288 m^3/s during the late summer survey (WSC 1985) and 21 m^3/s to 24 m^3/s in late winter (WSC 1986). Fish populations were sampled at 62 steep bank sites in late summer and 31 sites in late winter (Figure 2). Access to study sites was gained by use of a 5 m long jet-powered river boat. Sampling site selection was not based on a paired sample design involving rip-rap and natural banks of similar

hydraulic character and location. Fish sampling effort was distributed among bank (rip-rap and natural) and channel (main and side channel) types in both upper and lower

(a)

(b)

Figure 2. Natural bank (a) and rip-rap bank (b) study sites, North Thompson River, late summer.

river sections (Table 1). Late summer fish sampling was conducted with both beach seine and Gee minnow traps baited with frozen salmon roe (Bloom 1976). Minnow traps were set in 5 pairs (10 traps total), with 3 m – 4 m between pairs, along 15 m – 20 m of river bank at each sampling site. Traps were set over night with soak times of 16 h - 24 h. Beach seining data were not used because the method appeared to underestimate fish abundance at rip-rap relative to natural banks due to difficulty in sealing the lead line and preventing fish escape at rip-rap sites. In late winter, electrofishing without net enclosures, using a generator-powered DC electroshocker, was employed to provide fish population estimates for each study site. It was considered that under low winter flow and temperature conditions, when juvenile chinook and coho would be expected to hide within the substrate or wood debris cover (Chapman and Bjornn 1969; Bustard and Narver 1975), a net enclosure was not needed to prevent fish from leaving or entering the sampling area. The removal-depletion method (Seber and LeCren 1967), based on either a 2-catch or 3-catch strategy, was used to generate site population estimates by species.

Table 1. Distribution of North Thompson River fish sampling sites among bank and channel types by river section and season.

River section	Number of sites			
	Rip-rap bank	Natural bank	Main channel	Side channel
Late summer				
Upper river[a]	14	11	15	10
Lower river[b]	20	17	26	11
Totals	34	28	41	21
Late winter				
Upper river	6	3	7	2
Lower river	14	8	13	9
Totals	20	11	20	11

[a] Mainstem upstream of Blue River (Lempriere to Thunder River).
[b] Mainstem between Avola and McLure.

While minnow traps effectively captured juvenile coho in late summer, they caught few juvenile chinook or other fish species. Six trap efficiency tests, involving placement of known numbers of juvenile coho and chinook within net enclosures, were conducted at mainstem rip-rap (4) and natural bank (2) sites with low and moderate nearshore velocities (0-49 cm/s). These tests indicated that the minnow trap density and trapping duration employed in this study captured, on average, 53% of coho, and only 3% of chinook juveniles (Whelen and Lister 1985a).

Juvenile salmonids captured by each sampling method were identified as to species, then counted and released after size measures and scale sampling. Fork length measures to the nearest millimeter were obtained from live fish anesthetized with MS

222. Scale samples were obtained from a portion of the measured fish, to assist in interpreting length distributions relative to fish age.

Selected habitat parameters were documented at each 15 m – 20 m long fish sampling site. In the late summer survey, water depth (cm) and average water velocity (cm/s) were measured at 1 m intervals from 1 m to 5 m from shore at a representative point along the sampling site. Velocity at each measurement point was the average water column velocity estimated with a Marsh-McBirney Model 201 electromagnetic current meter set at 6/10 of depth from the water surface. Bank material size (D_{50} and D_{90}) and composition (Platts 1979), and quantities of overhanging vegetation and instream wood debris (percent of sampling site length) were also estimated. Sampling site length was the straight line distance between upstream and downstream limits as determined by measuring tape. In the late winter survey, habitat data collection at sampling sites was limited to visual categorization of bank type (natural or rip-rap) and velocity (low or moderate), and measurement of average water depth 1 m – 5 m off shore. Average surface water velocities within 1 m – 5 m from shore were estimated as either low (< 30 cm/s) or moderate (30 cm/s – 50 cm/s).

Juvenile salmonid densities are expressed as numbers of fish per 100 m of sampling site length. Statistical distributions of fish densities were skewed and lognormal (Gilbert 1987), with a small number of large values that tend to bias estimates of the mean. Zero density estimates were also common. Fish density data were therefore $\log_{10}(x + 1)$ transformed to normalize variances and facilitate statistical analysis (Green 1979). Mean fish densities cited in this report are geometric means calculated from log-transformed data.

Relationships between fish density and physical habitat variables were examined by linear correlation analysis. Predictive linear regression models (Sokal and Rohlf 1981) were also developed. Physical variables such as water velocity and wood debris, which also had lognormal distributions and included zero values, were $\log_{10}(x + 1)$ transformed for these analyses. Sample means were compared by two-sample T-test. Two-way analysis of variance (ANOVA) was used to examine fish density differences among bank and channel types and to test interaction of 2 factors (bank and channel type) relative to fish densities. A statistical probability (p) of < 0.05 was set as the limit for rejecting the null hypothesis (i.e., no significant difference or relationship between variables).

While juvenile chinook were the most abundant salmonid in the North Thompson River mainstem during late summer, concerns regarding non-representative sampling by beach seine, and inability to capture chinook by minnow traps, precluded use of chinook sampling data. Analysis of late summer North Thompson River data was therefore restricted to juvenile coho captures by minnow trap.

3.4.2 Results

Late Summer Survey: Two age groups of juvenile coho (age 0+ and 1+) were present in the study area. Age 0+ fish comprised the majority of coho in both upper and lower river study sections (Table 2). On average, lower river coho were younger and larger at a given age. Coho densities used in the following analyses represent combined samples of both age groups.

Table 2. Age composition and mean fork length at age of juvenile coho in upper and lower river study areas, North Thompson River, late summer. Minimum and maximum lengths in each sample are shown in brackets.

Study section	Sample size	Age 0+		Age 1+	
		Percent of sample	Length (mm)	Percent of sample	Length (mm)
Upper river (upstream of Blue River)	104	70.2%	52.0 (37-72)	29.8%	89.0 (69-108)
Lower river (downstream of Avola)	257	91.0%	56.6 (37-77)	9.0%	98.1 (73-108)

Juvenile coho densities and habitat parameters were highly variable and commonly non-normal in distribution (Figure 3). Mean coho densities were significantly higher at natural banks (16.1/100m) than rip-rap banks (6.8/100 m), (two-sample $t_{60} = 1.99$, $p = 0.05$, $n = 62$). Grouping of data by bank and channel type reveals marked differences in fish abundance and habitat parameters (Table 3). Natural banks and side channels supported higher coho densities and greater levels of riparian vegetation and wood debris cover. Statistical comparison of coho density differences among bank types (ANOVA, $F_{1,58} = 2.65$, $p = 0.11$) and channel types ($F_{1,58} = 3.92$, $p = 0.05$) indicates that channel type had greater influence on habitat selection by coho juveniles. This implies that both bank and channel type influence habitat selection by coho, but absence of a significant interaction effect (ANOVA, $F_{1,58} = 0.04$, $p = 0.76$) suggests that each factor acts independently. Given marked differences in coho density and habitat features at natural and rip-rap banks within side channels (Table 3), it would appear that site specific physical conditions, not channel type, influence habitat selection by coho. Side channels supported higher coho densities because favourable habitat, particularly lower velocity, was more common in side channel situations.

Nearshore water velocity had the clearest influence on habitat selection by juvenile coho (Figure 4). Coho density was negatively correlated with $\log_{10}$ velocity ($r^2 = 0.22$, $p = 0.001$, $df = 60$). At least 70% of coho at each bank type were captured at sites with relatively low velocity ($\leq$ 10 cm/s) that comprised 44% of sampling units. Riparian vegetation also had a positive effect on coho density (Figure 4), but its influence was more apparent at natural banks ($r^2 = 0.14$, $p = 0.05$, $df = 26$) than rip-rap ($r^2 = 0.07$, $p = 0.13$, $df = 32$). Wood debris supported relatively high coho densities when accumulations amounted to 10% - 20% of site length. Larger debris accumulations, exceeding 20% of site length, did not appear to be more beneficial than smaller accumulations (Figure 4).

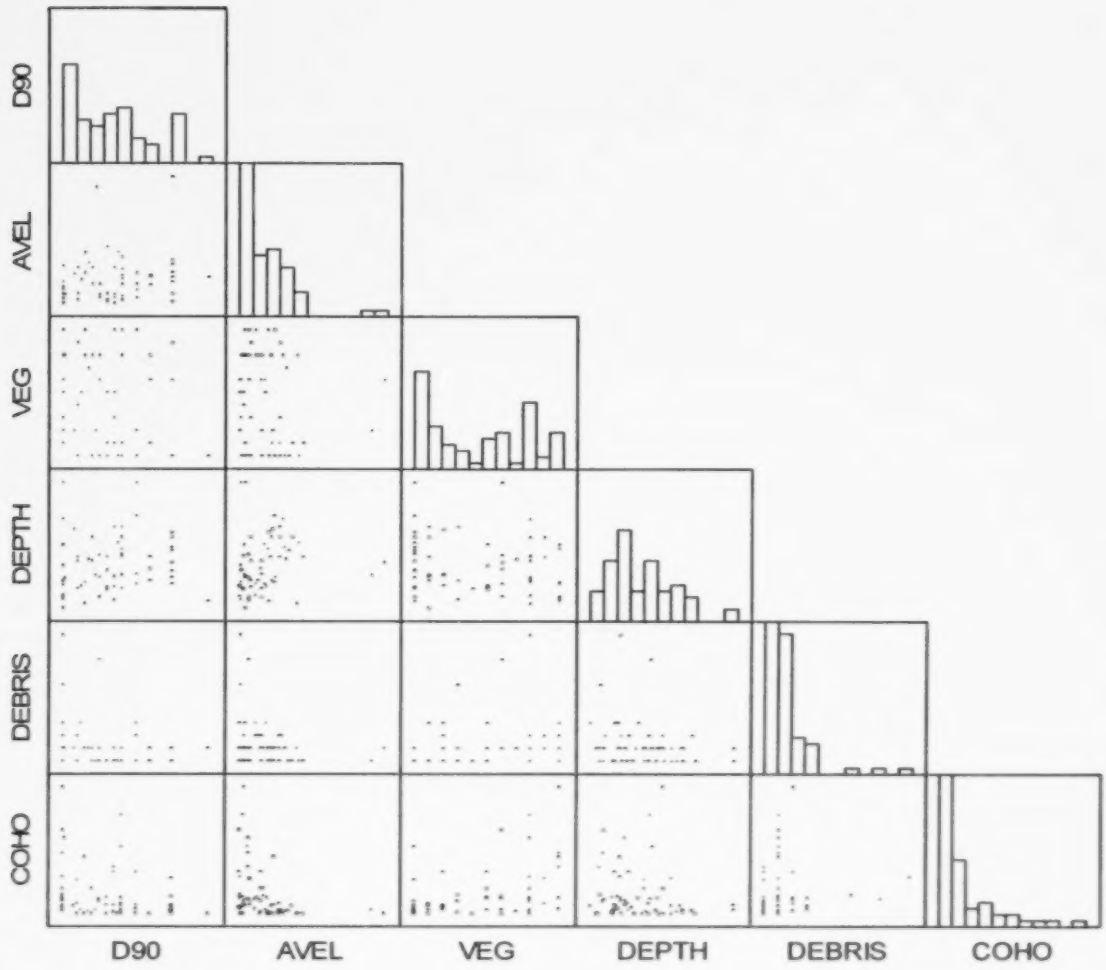


Figure 3. Scatterplot matrix showing relationships among juvenile coho density (COHO) and habitat variables, including bank material size (D90), nearshore water velocity (AVEL) and depth (DEPTH), bank vegetation (VEG) and wood debris (DEBRIS) at North Thompson River study sites in late summer.

Table 3. Mean values for juvenile coho density and selected habitat parameters, organized by bank and channel type, North Thompson River, late summer. Numbers in brackets represent 95% confidence intervals for mean density estimates.

	Main channel natural bank n = 16	**Main channel rip-rap bank** n = 25
Coho density (no./100 m)	**10.4** (4.0 – 11.9)	**5.5** (2.4-11.5)
Water velocity (cm/s)	24.3	20.5
Water depth (cm)	114.5	134.4
Riparian vegetation (%)	57.5	20.0
Wood debris (%)	15.0	6.0
	Side channel natural bank n = 12	**Side channel rip-rap bank** n = 9
Coho density (no./100 m)	**28.6** (14.7-55.1)	**12.2** (1.8-60.9)
Water velocity (cm/s)	7.7	30.4
Water depth (cm)	84.4	95.3
Riparian vegetation (%)	62.5	50.0
Wood debris (%)	22.5	8.9

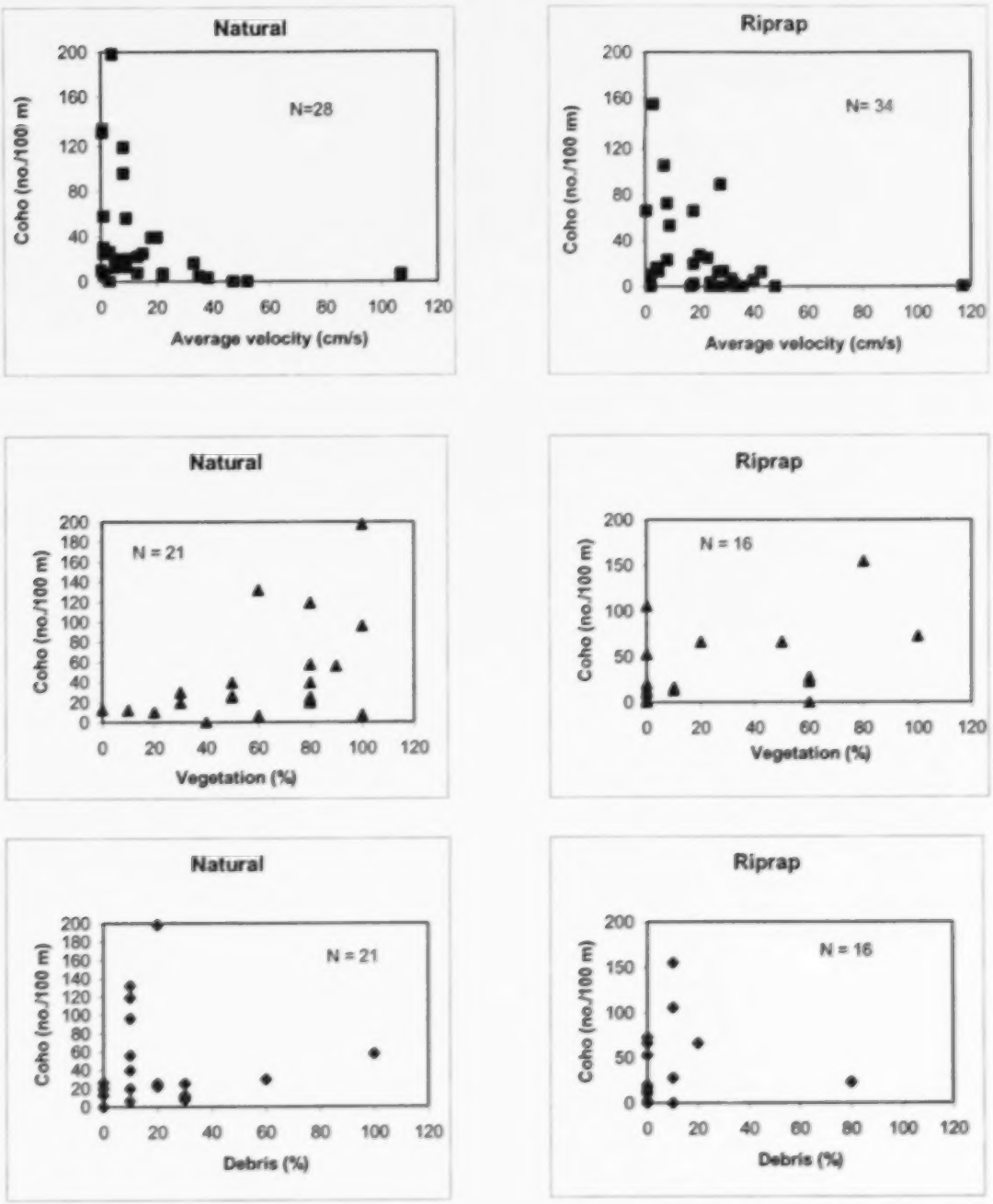


Figure 4. Juvenile coho density relative to nearshore water velocity, riparian vegetation and wood debris at natural and rip-rap bank sites, North Thompson River, late summer. As higher velocities tend to obscure the influence of other habitat features on habitat selection by coho, data from sites with velocity exceeding 20 cm/s were omitted from graphs showing coho density relative to vegetation and wood debris.

Habitat conditions at rip-rap tended to be less favourable for coho than conditions at natural banks. Sites with high velocity and an absence of riparian vegetation or wood debris were more common at rip-rap (Table 4). Juvenile coho were most abundant at sites with relatively low velocity and riparian vegetation that covered 50% or more of site length (Figure 5).

Table 4. Percentages of natural and rip-rap bank sampling sites with selected characteristics, North Thompson River, late summer.

Site characteristic	Natural banks (n = 28)	Rip-rap banks (n = 34)
Zero coho catch	7.1%	32.4%
Average velocity > 10 cm/s	42.9%	67.6%
Average depth > 100 cm	35.7%	64.7%
Riparian vegetation absent	3.6%	44.1%
Wood debris absent	21.4%	55.9%
Both vegetation and debris absent	3.6%	32.4%

Nearshore water velocity influenced coho density more than other habitat parameters, regardless of bank type. The large variance in scatterplots of coho density versus velocity (Figure 4) reflects not only velocity effects, but also the influence of other habitat factors specific to particular sites and bank types. Linear regression models fitted to coho density-velocity data for each bank type (Figure 4) should imply, by elevation of the regression line, the relative habitat suitability of a site as indicated by coho abundance. Regression models describing the coho density - nearshore velocity relationship at each bank type were as follows:

Natural bank: Coho density = 1.820 – 0.545 velocity
$r^2 = 0.26$, $p = 0.006$, $n = 27$ (one outlier excluded)

Rip-rap bank: Coho density = 1.921 – 0.839 velocity
$r^2 = 0.23$, $p = 0.004$, $n = 34$,

where: coho density = $\log_{10}$ (no./100 m +1) velocity = $\log_{10}$ (cm/s +1)

The models predict similar coho densities at low velocities but a progressive decline in suitability of rip-rap relative to natural banks as velocity increases (Table 5). The coho density-velocity relationship for rip-rap banks is largely influenced by a high incidence of zero density values (32.4%), particularly at high velocities.

At a subset of sampling sites with relatively favourable vegetation and nearshore velocity conditions, coho were more abundant at natural banks (mean 27.6/100 m) than rip-rap banks (mean 17.6/100 m), despite similarity in these key habitat parameters (Table 6). However, the difference in mean density at each bank type was not statistically significant ($t_{25} = 1.44$, $p = 0.13$, $n = 27$). While many rip-rap sites support coho densities comparable to those at natural sites, habitat suitability of natural banks for coho appears, on average, to exceed that of rip-rap banks in the North Thompson River during late summer.

Table 5. Regression model predictions of juvenile coho rearing densities at natural and rip-rap banks over a range of nearshore velocities, North Thompson River, late summer.

Average velocity (cm/s)	Predicted coho density (no./100 m)		Density ratio rip-rap:natural
	Natural bank	Rip-rap bank	
2	35.3	32.2	0.91
5	23.9	17.5	0.73
10	16.9	10.1	0.60
20	11.6	5.5	0.47
50	6.8	2.1	0.31

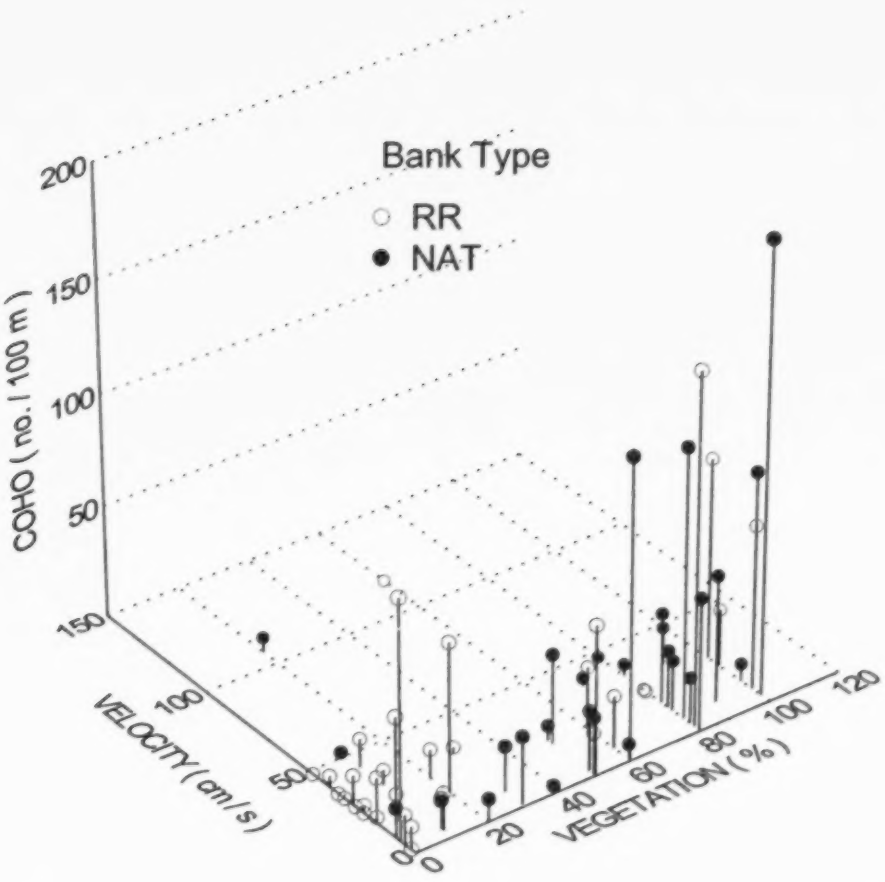


Figure 5. Juvenile coho density relative to nearshore velocity and riparian vegetation at natural (NAT) and rip-rap (RR) banks, North Thompson River, late summer.

Table 6. Mean values of coho density, nearshore velocity and riparian vegetation at selected natural and rip-rap sampling sites with relatively favourable vegetation (≥ 30% of site length) and velocity (≤ 25 cm/s) conditions, North Thompson River, late summer. Numbers in brackets represent 95% confidence intervals for mean density estimates.

	Natural banks	Rip-rap banks
No. of sites	19	8
Coho density (no./100 m)	27.6 (14.8 – 50.9)	17.5 (2.7 – 90.2)
Velocity (cm/s)	8.1	13.0
Vegetation (%)	67.9	73.8

Late Winter Survey: Though an attempt was made to include late summer sampling sites in the winter sample, snow conditions and ice cover made it difficult to gain access to many of the summer sites. As a result, only 10 of the 31 winter fish sampling units were at sites that had been sampled during the summer survey. Spot daytime water temperatures during the winter survey ranged from 0°C to 1.5 °C.

Chinook or coho juveniles were captured at 27 of 31 mainstem sampling sites during winter. Subyearling chinook, the most abundant salmonid, occurred only in the lower river section. Coho subyearlings and yearlings were encountered in both upper and lower river sections, but were more abundant in the upper river. Coho were present at 8 of 10 upper river sites and only 4 of 21 lower river sampling sites.

Two habitat variables, nearshore water depth and velocity category (low or moderate) were documented at each fish sampling site. Water depth was generally greater at rip-rap banks (mean depth 93 cm) than natural banks (mean depth 78 cm). There was no relationship between combined density of chinook and coho at study sites and nearshore water depth ($r^2 = 0.06$, $p = 0.20$, $df = 29$). Water velocity at study sites varied little within each study section (upper or lower river). All 10 upper river sites were in the moderate velocity category, whereas 20 of 21 lower river sites received a low velocity rating. This lack of variation and randomness in distribution of velocity conditions within and among study sections precluded correlation analysis.

Fish density differences between natural and rip-rap sites were not statistically significant in the case of chinook (two-sample $t_{20} = 1.05$, $p = 0.31$, $n = 22$), coho (two sample $t_{29} = 1.26$ $p = 0.22$, $n = 31$), or when the two species were combined (two sample $t_{29} = 1.88$, $p = 0.07$, $n = 31$) (Table 7). The larger size of bank material at rip-rap relative to natural banks (Table 8) may have been a factor in winter habitat selection.

Table 7. Mean densities of juvenile chinook and coho at natural and rip-rap banks, North Thompson River in winter. Ninety-five percent confidence intervals are shown in brackets.

Species	Natural banks		Rip-rap banks	
	No. of cases	Density (no. /100 m)	No. of cases	Density (no./100 m)
Chinook[a]	8	**9.9** (3.3 - 26.4)	14	**19.0** (7.9 - 44.2)
Coho	11	**0.5** (0.1-1.5)	20	**1.7** (0.4 – 4.2)
Species combined[b]	11	**6.7** (2.4 – 16.2)	20	**17.7** (9.2 – 33.4)

[a] Smaller sample size for chinook reflects the fact that chinook were present only in the lower river section.

[b] Fish density at a particular sampling site was treated as the sum of chinook and coho densities.

Table 8. Estimated bank D_{90} at natural and rip-rap bank sampling sites, North Thompson River, based on data from late summer survey.

		Bank D_{90}(cm)		
Bank type	No. of cases	Mean	Minimum	Maximum
Natural	28	24.9	0	150
Rip-rap	34	93.2	15	200

3.5 THOMPSON RIVER

3.5.1 Methods

Field work at Thompson River was conducted in 1987 at high (May 21 – June 10 and June 22 – July 2) and medium (August 22 – September 10) flow levels , and during March 3-9, 1985 under winter low flow conditions. River discharge, estimated at Spences Bridge gauge, averaged 1600 m^3/s at high flow, 450 m^3/s at medium flow and 175 m^3/s during the winter survey (WSC 1985, 1988). Ranges in spot daytime water temperatures during high flow, medium flow and winter surveys were 13° – 17°C, 17° – 19°C, and 0.5 - 1°C, respectively.

During summer field work, salmonid densities were documented along steep banks (1.5 h to 1 v or steeper) with no vegetative cover and composed of either natural cobble-boulder material, small rip-rap ($D_{50} \leq 30$ cm) or large rip-rap ($D_{50} > 30$ cm) (Figure 6). Access to study sties was gained by use of a 5.5 m long jet-powered river boat. Study design involved paired comparisons of fish density at sites in the 3 bank categories (Table 9). Sites in a given pair were generally situated within 3 km and sampled within the same 24 h period. As nearshore velocity was known to influence habitat selection by juvenile chinook and rainbow trout in the Thompson River (Beniston et al. 1985), each site of a particular bank type was paired with one of another bank type in the same velocity category (Table 10). Fish were enumerated visually by swimmer equipped with dry suit, mask and snorkel (Schill and Griffith 1984). A single observer, moving downstream, counted numbers of fish by species within 3 m of the river bank in the May – June surveys and within 5 m of the bank in the August – September survey. Population estimates for a given site were based on maximum numbers of fish observed in 2 passes. Site length varied from 25 m to 100 m. Bank length surveyed in this manner totalled 3.5 km in May – June and 5.6 km in August – September.

Table 9. Numbers of paired comparisons of juvenile salmonid densities by bank type, Thompson River, 1987.

Survey period	Large rip-rap versus cobble	Large rip-rap versus small rip-rap	Small rip-rap versus cobble
May 21 – June 10	12	--	--
June 22 – July 2	10	9	8
August 22 - September 10	17	12	10

(a)

(b)

(c)

Figure 6. Natural cobble-boulder bank (a), small rip-rap bank (b), and large rip-rap bank (c) study sites, Thompson River, at low flow in April.

Table 10. Nearshore water velocity criteria used in paired site comparisons, Thompson River, 1987.

Velocity rating	Criteria
High	Surface velocity of 50 cm/s located less than 2 m from bank.
Moderate	Surface velocity of 50 cm/s located 2 m or more from bank. Average water column velocity 1 m – 5 m from bank is 10 cm/s – 47 cm/s.
Low	Surface velocity of 50 cm/s is more than 5 m from bank. Average water column velocity 1 m – 5 m from bank is less than 10 cm/s.

Habitat parameters were documented at each fish enumeration site during summer surveys. Rock size along the water line was visually estimated at the time of each fish population census, because rock size varied with water level due to vertical sorting which resulted in larger material at the bank toe. Nearshore water depth and velocity measures were also obtained in the manner described for the North Thompson River study.

Late winter fish sampling was conducted at 23 sites, including 9 large rip-rap, 4 small rip-rap and 10 cobble-boulder sites totalling 430 m long. A generator-powered DC electroshocker without net enclosure was used to provide a fish population estimate for each site, based on a 2-step removal method (Seber and LeCren 1967). As in the case of the North Thompson River winter study, a net enclosure was not considered necessary at sampling sites because of low water velocities and the tendency for chinook and rainbow trout juveniles to hide in the substrate at low winter temperatures (Chapman and Bjornn 1969). At each sampling site, bank rock size was visually estimated, nearshore water depth was measured at 3 m from shore, and site velocity conditions were characterized as either high (50 cm/s surface velocity < 2 m from shore) or low (50 cm/s surface velocity $\geq$ 2 m from shore).

Fish length measures and scale samples were obtained by seine and electrofishing in summer, using the same handling and measurement procedures described for the North Thompson River study.

Fish density data (number per 100 m of bank length) were $\log_{10}(x + 1)$ transformed to normalize variances and facilitate statistical analysis which was based on geometric mean fish densities calculated from the transformed data. Paired comparisons of fish densities at rip-rap and natural banks in summer were analyzed with the non-parametric Wilcoxon signed-ranks test (Sokal and Rohlf 1981) because of non-normal distribution of the data. Relationships between winter fish density and nearshore water depth were examined by linear correlation analysis. Differences in fish densities by bank type during winter were compared by one-way ANOVA. Fish density differences relative to velocity class (low or moderate) in winter were analyzed by two-sample T-test.

3.5.2 Results

Early and Late Summer Surveys: Chinook subyearlings and rainbow trout parr were the most common and abundant fish species at sampling sites in all study periods. Mean lengths of these species increased between early and late summer surveys (Table 11).

Table 11. Mean fork lengths of juvenile chinook subyearlings and rainbow trout parr (age 1+ and 2+) in the Thompson River during summer.

Sampling period	Chinook 0+			Rainbow parr		
	Sample size	Mean length (mm)	Length range (mm)	Sample size	Mean length (mm)	Length range (mm)
May 21 – June 10	715	49	31-70	59	101	70-180
June 22 – July 2	337	62	45-95	28	108	87-170
September 3-20	253	74	55-96	13	168	129-210

Rock size at each bank type increased between early and late summer, due to declining water level and vertical sorting of rock by size. Over the summer period large rip-rap remained the largest bank material and cobble-boulder the smallest (Table 12).

Table 12. Estimated mean rock size at Thompson River study sites at high flow in June and medium flow in September. Sample sizes are given in Table 9.

Sampling period	Bank D_{50}(cm)[a]			Bank D_{90} (cm)[a]		
	Large rip-rap	Small rip-rap	Natural cobble-boulder	Large rip-rap	Small rip-rap	Natural cobble-boulder
June	57	17	12	79	27	17
September	61	22	14	102	61	22

[a] Rock size estimated at water's edge at time of survey.

Large rip-rap supported significantly higher chinook and rainbow parr densities than cobble-boulder banks (Wilcoxon signed-ranks test, $p = 0.03$, $n = 11$) during the initial high flow survey (May 21-June 10) (Table 13). In the second high flow survey during late June, there were no differences between chinook and rainbow parr densities (Wilcoxon signed-ranks test, $p > 0.15$) (Figure 7). Differences between surveys in relative densities of chinook at large rip-rap and cobble-boulder banks may have been related, at least in part, to a change in fish size between survey periods (Table 11).

Table 13. Mean chinook subyearling and rainbow trout parr densities at paired large rip-rap and natural cobble-boulder banks, Thompson River, May 21 – June 10. Asterisks indicate significantly higher density at large rip-rap relative to cobble-boulder ($P< 0.05$).

Species	Mean density (no./100 m)	
	Cobble – boulder n = 11	Large rip-rap n = 11
Chinook 0+	15.5* (5.2 – 43.2)	106.2* (67.6-177)
Rainbow parr	0.1* (0.01 – 0.4)	1.7* (0.4-3.9)

By September, differences in fish use of larger rock relative to smaller rock became more pronounced except in the case of chinook at small rip-rap and cobble-boulder banks (Figure 7). Chinook densities at large rip-rap were significantly higher than densities at cobble-boulder (Wilcoxon signed-ranks test, $p = 0.001$, $n = 10$) and small rip-rap ($p = 0.01$, $n = 9$). The moderately higher chinook density at cobble-boulder relative to small rip-rap sites was not significant ($p = 0.78$, $n = 8$). Rainbow trout parr densities at large rip-rap significantly exceeded those at cobble-boulder sites (Wilcoxon signed-ranks test, $p = 0.01$, $n = 11$). Mean parr density did not differ between large and small rip-rap sites ($p = 0.11$, $n = 9$) or between small rip-rap and cobble-boulder banks ($p = 0.41$, $n = 8$). The greater influence of rock size on habitat selection in September, relative to June, may have been related to increased rock size at the water line (Table 12), which likely altered the nearshore hydraulic environment. Larger fish size (Table 11) may also have enabled chinook and rainbow trout juveniles to utilize a wider range of habitat conditions.

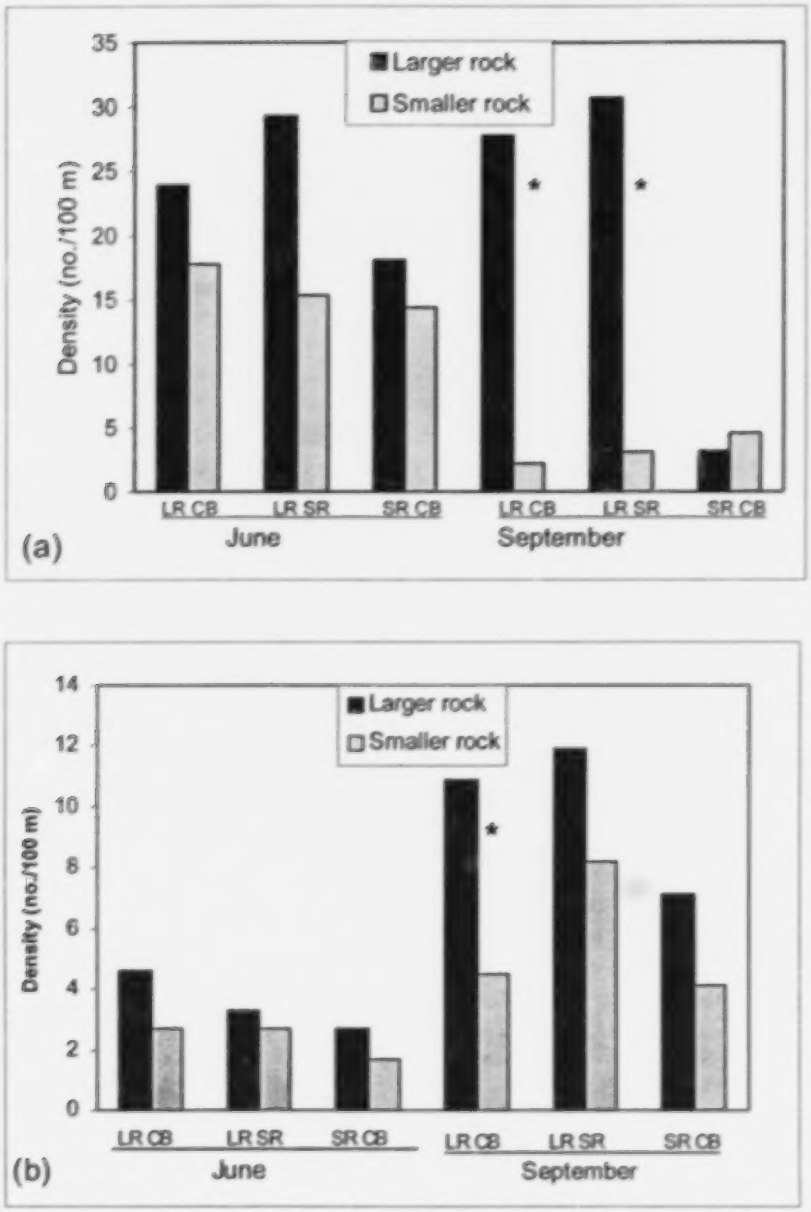


Figure 7. Mean juvenile chinook subyearling (a) and rainbow trout parr (b) densities at large ripirap (LR), small rip-rap (SR) and cobble-boulder (CB) banks, Thompson River, late June and September. Data are from paired sample comparisons among bank types. Asteriks indicate that the means were significantly different ($p < 0.05$)..

Late Winter Survey: A limited fish sampling effort was undertaken at each bank type during late winter. Chinook yearlings were more abundant at large rip-rap than at either small rip-rap or natural cobble-boulder banks (ANOVA, $F_{2,20} = 3.58$, $p = 0.05$) (Table 14). Rainbow parr density did not differ among bank types (ANOVA, $F_{2,20} = 1.44$, $p = 0.26$).

Observed differences in fish density among bank types may have been related, in part, to other habitat features (Table 14). Chinook (two-sample $T_{21} = 1.80$, $p = 0.08$, $n = 23$) and rainbow parr (two-sample $T_{21} = 0.72$, $p = 0.49$, $n = 23$) habitat selection was not influenced by nearshore water velocity at sampling sites. As velocity conditions were similar at the 2 main bank types, cobble-boulder and large rip-rap (Table 14), it would seem unlikely that velocity had a significant effect on habitat selection by chinook. The association of chinook density with bank type and mean water depth (Table 14) was supported by a relatively strong correlation between density and depth at individual sampling sites ($r^2 = 0.29$, $p = 0.01$, $df = 21$). Though rainbow parr density and mean water depth were also highest at large rip-rap, no measurable relationship was evident between rainbow density and water depth at individual sites ($r^2 = 0.01$, $p = 0.71$, $df = 21$). In the case of juvenile chinook, therefore, selection of large rip-rap as a substrate for over-wintering may have been influenced by the greater water depth associated with that bank type.

Table 14. Mean juvenile chinook and rainbow trout parr densities, and water depth and velocity at natural, cobble-boulder and rip-rap banks, Thompson River, late winter. Numbers in brackets show 95% confidence intervals for mean density estimates. Asterisks indicate significant difference in fish density among bank types ($p < 0.05$).

	Cobble-boulder banks	Small rip-rap banks	Large rip-rap banks
No. of cases	10	4	9
Mean chinook density (no./100 m)*	0.8* (0-4.0)	1.7* (0-16.4)	8.0* (1.4-32.8)
Mean rainbow parr density (no./100 m)	0.9 (0-3.2)	0.7 (0 - 2.9)	2.6 (0-12.4)
Mean water depth (cm)	90.5	102.5	122.2
Percent of sites in low velocity category	60%	75%	56%

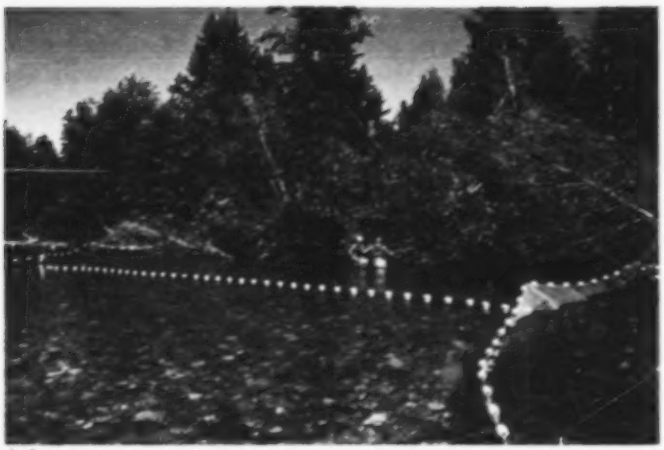
3.6 COLDWATER RIVER

3.6.1 Methods

Studies at Coldwater River involved 6 paired comparisons of juvenile salmonid densities at a site recently altered by highway construction and an unaltered reference site (Figure 8). The dominant feature of altered study sites was the extensive rip-rap protection on one or both banks of the stream. Rip-rap was keyed into the stream bed to prevent undermining by scour. Relatively large rip-rap, with mean D_{50} of 59 cm and mean D_{90} of 99 cm, was used to enhance fish habitat (M. Miles and Associates Ltd. 1992). Reference sites had unaltered, vegetated banks that, in some cases, had been affected by past forest harvest. Field work was conducted under low flow conditions in late summer and late winter of 3 study years. While juvenile chinook, coho and steelhead were the focus of the study, limited data relating to mountain whitefish and bull trout are also presented.

Late summer field work was conducted over a 3 week period during August 20 – September 20 of 1986, 1987, and 1988 at average stream discharges of 0.7 m^3/s–0.9 m^3/s (Brookmere gauge; WSC 1987, 1988 and 1989). Daytime water temperatures ranged from 8°C to 20°C. Electrofishing within net enclosures, using a generator-powered DC electroshocker, was employed to provide fish population estimates at each study site (Beniston et al. 1988). The removal-depletion method (Seber and LeCren 1967), based on either a 2-catch or 3-catch strategy, was used to generate population and density estimates by species. As the majority of juvenile salmonids were concentrated near rip-rap or natural banks along the outside bend (Beniston et al. 1988), site fish populations likely reflected the influence of bank conditions.

Late winter surveys required 10 days of field work during February 27 to March 30 in 3 study years (1988-1990). Each annual survey was scheduled to occur immediately after ice-out, and therefore varied in timing from year to year. River discharges averaged 1.2 m^3/s – 1.6 m^3/s and spot daytime water temperatures were generally 1°C - 5°C. As most salmonid juveniles occupied bank habitat along the outside of the river bend during winter, water depth and bank cover were major determinants of habitat suitability (Beniston et al. 1988). Fish sampling was therefore concentrated in a 3 m - 4 m band along the outside bank. The stratified sampling design involved comparison of fish densities in 10 m – 13 m long sampling units at a rip-rap sampling unit paired with a nearby natural bank unit having either high or low cover rating and comparable water depth and velocity. At sampling units with high cover rating, $\geq$ 9% of sampled area consisted of wood debris, undercut bank, overhead cover or instream vegetation. Units in the low cover category had < 9% of area with such cover. Paired sampling units were also selected to ensure equal numbers of deep (> 60 cm) and shallow ($\leq$ 60 cm) sites at both rip-rap and natural banks. An effort was made to select pairs of sampling sites that were similar in nearshore water depth and velocity, as indicated by the winter, 1989 example in Table 15.



(a)

(b)

(c)

Figure 8. Natural banks with high cover (a) and low cover (b) rating, and rip-rap bank (c), Coldwater River, late summer.

Table 15. Physical characteristics of paired natural and rip-rap bank sampling units, Coldwater River mainstem, late winter, 1989.

Site type	Habitat type	Mean value[a]			
		Depth[b] (cm)	Velocity[b] (cm/s)	Rock cover (%)	Vegetation or wood debris %)
Deep[b] (> 60 cm)	Rip-rap	65	15	26	0.4
	Natural (high cover)[c]	69	16	3	22
	Natural (low cover)[c]	58	17	5	1
Shallow[b] (≤ 60 cm)	Rip-rap	33	27	23	0
	Natural (high cover)	36	32	5	13
	Natural (low cover)	33	33	6	0.5

[a] Based on 8 sampling units of each habitat type.
[b] Depth and velocity measurements taken 1 m – 3 m from shore.
[c] High cover = ≥ 9% of area; low cover = < 9% of area.

Mountain whitefish and bull trout were present in relatively low numbers at Coldwater River study sites. Populations of these species were estimated at 14 pairs of mainstem altered and reference sites in early summer, late summer and late winter over a 2-year period. Twelve site pairs were from late summer sampling.

Habitat features of each fish sampling site were documented in the manner described for the North Thompson River study. Estimated area of rock cover or non-rock cover (overhanging vegetation or wood debris) in each sampling unit was also noted. Fish size and age data were obtained in the manner described for the North Thompson River study.

Summer fish densities are expressed as number per 100 m^2 of stream area, whereas winter fish densities are expressed as number per 100 m of bank length. Fish density data were $\log_{10}$ (x + 1) transformed to normalize variance. Statistical analysis was based on geometric mean fish densities calculated from the transformed data. Paired comparisons of fish densities at altered and reference sites or rip-rap and natural bank sampling units were analyzed with the non-parametric Wilcoxon signed-ranks test (Sokal and Rohlf 1981) because of non-normal distribution of data.

3.6.2 Results

The Coldwater River mainstem data set includes 5 salmonid species and several age groups in the case of steelhead, bull trout and whitefish, as shown by the example in Table 16. The coho population included wild and planted hatchery-reared subyearlings. Wild coho were treated as one age group (age 0+), though a small portion (1.5%) were yearlings (age 1+). Most steelhead parr were age 1+ and 2+ fish, but a small proportion

(2.6%) were age 3+ to 5+. With respect to bull trout parr, age 1+ fish comprised 67% of the sample, with the remainder being age 2+ or 3+. Ages of mountain whitefish adults were not determined.

Table 16. Mean fork length and length range of juvenile salmonids by sampling season, Coldwater River mainstem, 1987-1988. Data are from Beniston et al. (1988).

Species and age group	Late summer			Late winter		
	Sample size	Mean length (mm)	Length range (mm)	Sample size	Mean length (mm)	Length range (mm)
Chinook 0+	1539	62	39-90	181	73	50-94
Wild coho 0+	1294	47	32-61	153	59	43-73
Hatchery coho 0+	152	80	56-110	12	82	75-90
Steelhead 0+	2931	46	30-68	658	57	40-76
Steelhead parr	1058	99	69-230	20	105	78-165
Bull trout 0+	7	55	41-69	2	76	74-77
Bull trout parr	50	117	80-197	6	158	145-200
Mountain whitefish 0+	51	58	48-88	12	92	85-100
Mountain whitefish adult	44	296	230-360	----	----	------

Late Summer Survey: In late summer, juvenile chinook (Wilcoxon signed-ranks test, $p = 0.52$, $n = 18$) and wild coho ($p = 0.35$), did not differ between rip-rap banks and reference sites. Hatchery coho juveniles were present in higher densities at reference sites ($p = 0.02$). Both steelhead 0+ ($p = 0.01$) and steelhead parr ($p = 0.01$) were more abundant at rip-rap sites (Figure 9).

Late Winter Survey: Habitat selection in winter illustrated the importance of bank cover and water depth. Chinook, coho and steelhead 0+ juveniles were all more abundant at deep, natural sites with high cover rating than at deep rip-rap sites (Figures 10 and 11). Steelhead parr densities, however, were higher at deep rip-rap than deep natural banks with high cover. In the case of coho (wild and hatchery) and steelhead parr, these

differences were statistically significant (Wilcoxon signed-ranks test, $p < 0.05$, $n = 22$). Deep rip-rap banks with low cover supported higher densities of steelhead parr than deep natural banks with low cover, (Wilcoxon signed-ranks test, $p < 0.05$, $n = 22$). Shallow sites carried low densities of chinook and coho, regardless of bank type or cover. Steelhead were relatively more abundant at shallow sites. Shallow rip-rap supported lower steelhead 0+ densities than shallow natural banks with high cover, but significantly higher densities of steelhead parr (Wilcoxon signed-ranks test, $p < 0.05$, $n = 20$).

Other Species: Adult whitefish were more abundant at reference sites than rip-rap sites (Wilcoxon signed-ranks test, $p = 0.01$, $n = 14$). (Figure 12). Bull trout parr and yearlings and whitefish yearlings did not differ in density at altered and reference sites.

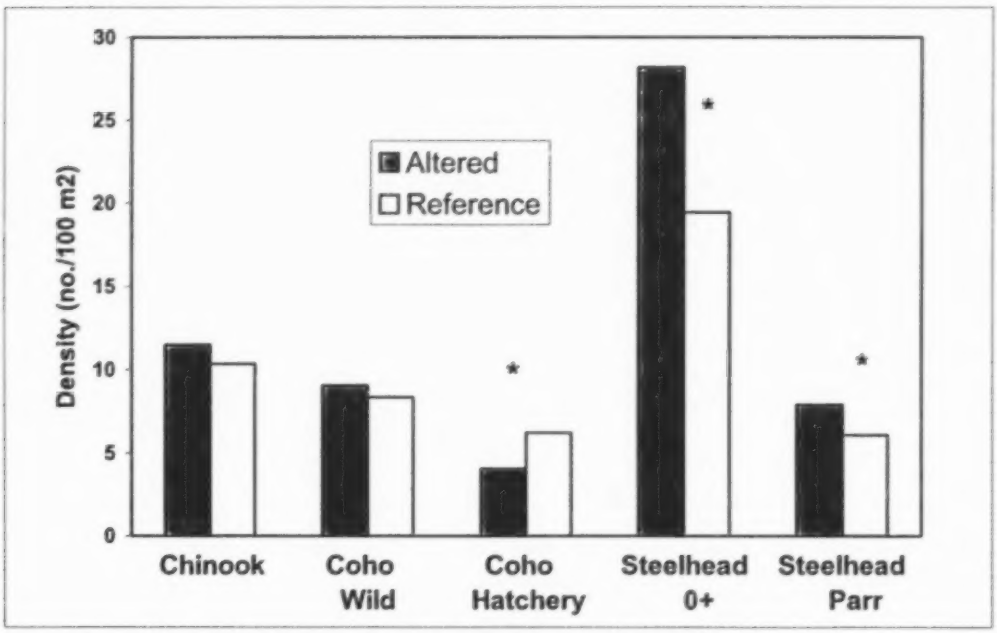


Figure 9. Mean juvenile salmonid densities at paired altered and reference sites, Coldwater River mainstem, late summer. Asteriks indicate that the means were significantly different ($p < 0.05$).

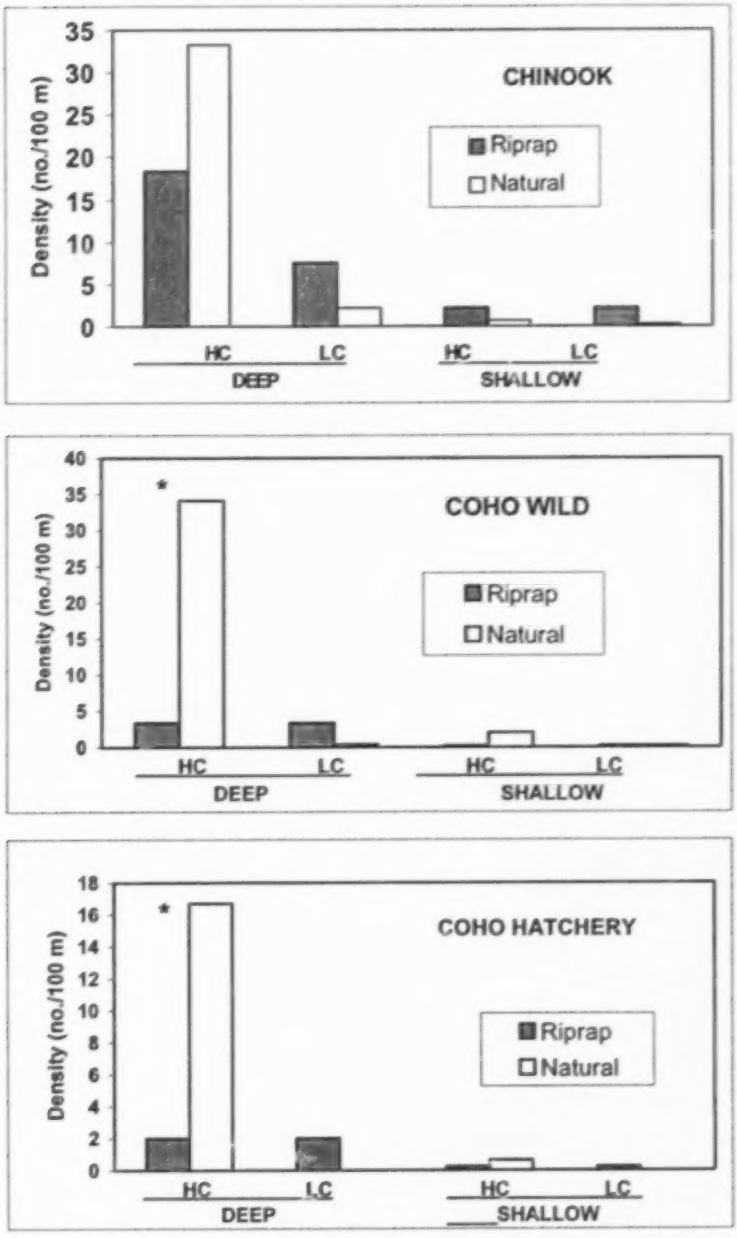
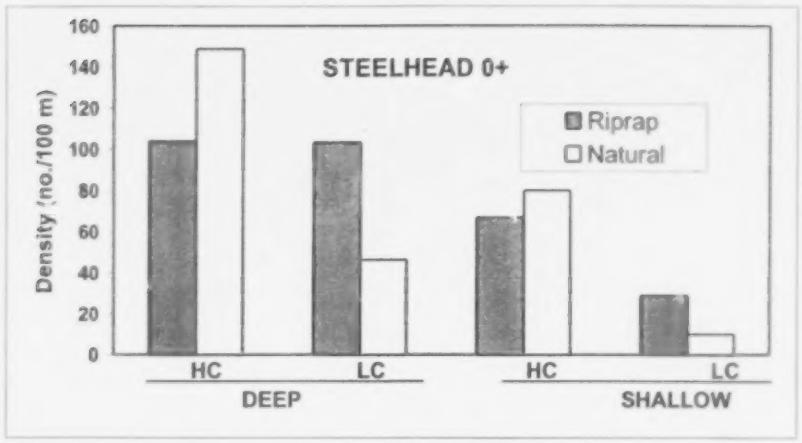


Figure 10. Mean juvenile chinook and coho densities at paired natural and rip-rap bank sites relative to water depth (deep and shallow) and high (HC) or low (LC) bank cover, Coldwater River mainstem, late winter. Asteriks indicate that the means were significantly different ($p < 0.05$).



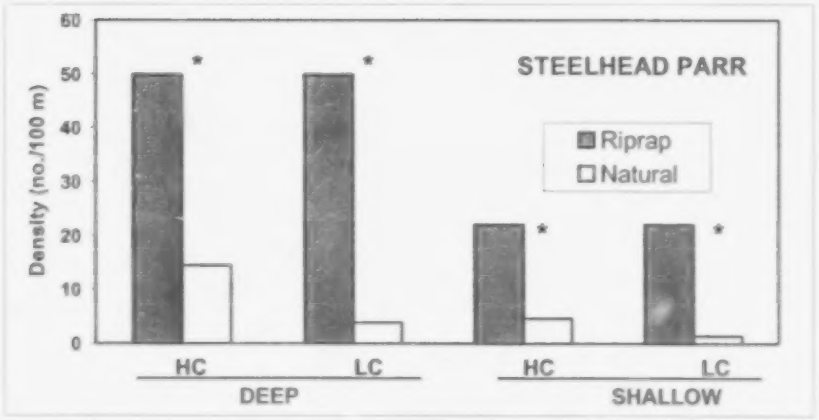


Figure 11. Mean juvenile steelhead 0+ and parr densities at paired natural and rip-rap bank sites relative to water depth (deep and shallow) and high (HC) or low (LC) bank cover, Coldwater River mainstem, late winter. Asteriks indicate that the means were significantly different ($p < 0.05$).

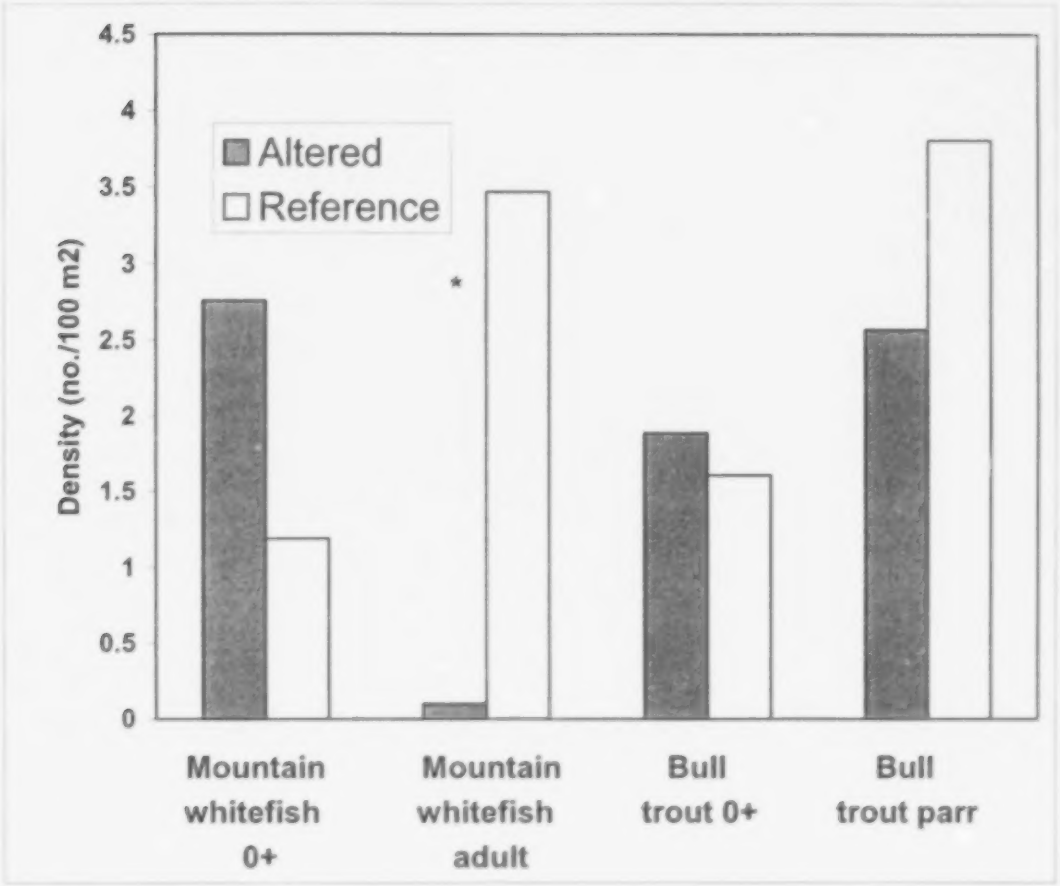


Figure 12. Mean mountain whitefish and bull trout densities at Coldwater River mainstem altered and reference sites. Adult mountain whitefish were ≥ 230 mm in length. Asteriks indicate that the means were significantly different ($p < 0.05$).

3.7 LOWER FRASER RIVER

3.7.1 Methods

Fish sampling in the lower Fraser River was conducted during 4 spring and summer surveys (April 29 – August 23, 1985) and one winter survey (January 28 – February 7, 1986) over a broad range of river discharge and water temperature conditions (Table 17). Study sites included steep, natural and rip-rap banks along the railway embankment on the south bank of the Fraser River mainstem (Figure 13). Sampling units numbered 19 in late April – early May, 46 in June, 50 in July, 32 in August, and 26 during the winter survey. Field work in May was confined to sites in the lower study section (Sumas River to Mission), while the June, July, August and winter surveys included sampling sites in both upper Hope to Agassiz) and lower sections.

Table 17. Average discharge, temperature and water level of the lower Fraser River during field surveys in 1985 and 1986.

Survey period	Discharge (m^3/s)[a]	Water level (m)[b]	Temperature (°C)[c]
April 29-May 4, 1985	2248	1.38	6.3
June 5-22, 1985	8048	5.07	11.7
July 9-17, 1985	4914	2.92	15.8
August 19-23, 1985	2470	1.38	15.7
January 28 – February 7, 1986	805	0.97	3.0

[a] Discharge at Hope gauge (WSC 1986, 1987).
[b] Water level at Mission gauge (WSC 1986, 1987).
[c] Average daytime spot water temperatures at Hope (WSC, Vancouver, B.C., unpublished data).

(a)

(b)

(c)

Figure 13. Natural bank in August (a) and vegetated rip-rap site at high flow in June (b) and at lower flow in August (c), lower Fraser River.

Spring and summer fish sampling was conducted with a 5 m long jet-powered river boat. Fish were captured by electrofishing within a 20 m long net enclosure at sites with banks of coarse, natural rock or rip-rap, and by beach seine (20 m or 6 m long) at sites with relatively fine bank and bottom material in the silt-gravel size range. To prevent fish escaping under the enclosure net at sites with rock or rip-rap substrate, where electrofishing was employed, an attempt was made to seal the net's lead line on relatively smooth bottom material at the bank toe. With each sampling method, site population estimates were derived by the removal-depletion method using a 2- or 3-step strategy (Seber and LeCren 1967).

In the course of spring and summer fish sampling, it was apparent that in the turbid Fraser River water electrofishing efficiency was probably lower than that of seining, because a portion of fish stunned by the electroshocker may have remained on the river bottom rather than being attracted to the anode at the water surface (Reynolds 1983). Since electrofishing was employed mainly at sites with large substrate such as rip-rap, the net used to enclose a sampling area could not be drawn into shore to capture fish remaining after electrofishing was completed. Only those fish which came to the water surface during electrofishing were available for enumeration. Tests were conducted on July 2 – 3, 1986 to determine how much the electrofishing technique employed in 1985 spring-summer surveys might have underestimated juvenile chinook densities (Beniston et al. 1986). Twenty-four sites in the study area were selected for this sampling evaluation. The test sites were steep banks of relatively fine substrate (sand-silt) which could be effectively electrofished and then seined. At each site a 20 m long seine was set 3 m off shore and the area enclosed by the net was electrofished with 2 or 3 passes through the enclosed area. After electrofishing, the seine was drawn into shore to enumerate chinook remaining in the enclosure. For each test site, a removal-depletion fish population estimate was compared to the estimated total site population as follows:

$$\text{Total site population} = \text{total fish captures from all electrofishing passes} + \text{fish captures in seine net}$$

$$\text{Electrofishing efficiency} = \frac{\text{electrofishing site population estimate}}{\text{total site population}}$$

These tests indicated that electrofishing removal-type estimates represented, on average, 0.70 of the actual population at sites sampled by that method (Table 18). Juvenile chinook population estimates derived for sites sampled by electrofishing in the spring-summer survey were therefore multiplied by 1.43 (1.0/0.7) to correct for reduced capture efficiency. Though fish sampling during winter (Table 17) employed the multiple-pass electrofishing method used at rip-rap sites in spring and summer, no correction factor was applied to winter population estimates because water clarity was high during that period.

Table 18. Results of tests to estimate electrofishing capture efficiency of chinook juveniles at lower Fraser River sampling sites during spring and summer.

No. of tests	Electrofishing capture efficency			
	Mean	Standard deviation	95% confidence interval	Range
24	0.70	0.15	0.64 - 0.76	0.39 - 1.0

Habitat features at fish sampling sites were recorded during each survey using a protocol similar to that employed at North Thompson River. At a representative point along each sampling unit, bank material size (D_{50} and D_{90}) and vegetative bank cover within 3 m of the wetted river edge (percent of site length with grasses or overhanging brush and trees) were documented. Water depth and average water column velocity were also measured at a point 3 m from the bank.

Field work was also conducted in the Fraser River mainstem during late January and early February to assess juvenile salmonid over-wintering at rip-rap banks relative to natural banks (Table 17). Fish sampling was conducted at 12 natural and 14 rip-rap bank sites distributed over the upper and lower study sections. Electrofishing with net enclosure was used to develop 2- or 3-step removal-depletion estimates (Seber and LeCren 1967) of salmonid populations at each rip-rap and some natural bank sampling sites. A 20 m long beach seine was used to sample fish at most natural bank sites. Due to the high water clarity in winter, it was not considered necessary to apply a correction factor to electrofishing population estimates. Habitat character at sampling sites, particularly water depth, velocity, and bank material size (D_{50}), was documented in the same manner as in the spring-summer survey.

Captured juvenile salmonids were enumerated at each site and a sample of up to 20 live, anesthetized fish of each salmonid species was measured (fork length to nearest mm). Scale samples were collected from representative subsamples to facilitate interpretation of length frequency data.

Fish density data (number per 100 m of bank length) were $\log_{10}(x + 1)$ transformed to normalize variances and facilitate statistical analysis, which was based on geometric mean fish densities calculated from the transformed data. Bank vegetation and velocity data were also log-transformed to normalize variances. Relationships between fish density and habitat variables were examined by correlation analysis. Because of low salmonid abundance during winter, densities of individual species were combined for statistical analysis. Differences in fish densities at rip-rap and natural banks during a given sampling period were compared with the two-sample T-test (Sokal and Rohlf 1981). For the June to August period, two-way ANOVA was used to test changes over that period in chinook densities relative to bank type and nearshore velocity, based on significance of the interaction between time (month) and either bank type or velocity category.

3.7.2 Results

The focus of this study was subyearling (age 0+) chinook which were the most abundant and widespread fish species at Fraser River mainstem sampling sites. Few yearling chinook were present. Age 0+ chinook captured in the May survey likely originated from the ocean-type (Healey 1991) Harrison River population, the only significant lower Fraser River chinook stock (Fraser et al. 1982). Department of Fisheries and Oceans downstream migrant trapping at Mission shows that in the year of study (1985) the major portion of Harrison River chinook fry migration in the Fraser mainstem occurred during April 10 – May 20 (Beniston et al. 1986). Chinook juveniles captured in this study during June and subsequent surveys are believed to have been from stream-type chinook populations that spawn in the Thompson River and upper Fraser River systems. Juvenile fish sampling in the Fraser River mainstem near Hope in 1985 indicated that chinook fry from sources upstream of Harrison River were present in that area by May 20 (Zallen and Boyd 1986). Mean length of chinook subyearlings in the study area increased from 40 mm in May to 64 mm in August (Table 19).

Table 19. Mean length and length range of chinook subyearlings by sampling period, lower Fraser River, 1985.

Sampling period	Sample size	Mean length (mm)	Length range (mm)
May	100	40.3	35 – 47
June	336	43.7	33 – 60
July	289	55.1	32 – 77
August	55	64.4	52 – 83

Spring-Summer Surveys: Juvenile chinook densities were lower at rip-rap than natural banks in 3 of 4 spring and summer sampling periods (Table 20). Rip-rap sites tended to have greater water depth and nearshore velocity, and supported less riparian vegetation. Estimated mean D_{50} of rip-rap and natural bank types varied among survey periods, with rip-rap (mean 25 cm – 54 cm) being larger than either cobble-gravel (mean 4 cm – 12 cm) or sand-silt (mean < 1 cm) banks (Beniston et al. 1986).

In May, chinook abundance did not differ statistically between natural and rip-rap banks (two-sample $t_{17} = 0.41$, $p = 0.69$, $n = 19$). Chinook density was not related ($r^2 < 0.07$, $p > 0.30$, $df = 17$) to measured habitat parameters. Most study sites had a zero vegetative cover rating in May because of low river levels (Table 17) relative to elevation of the vegetation trim line. The tendency for higher chinook densities at natural banks may have been related to lower depths and velocities at those sites (Figure 14), in keeping with habitat preferences of recently emerged ocean-type chinook fry (Lister and Genoe 1970).

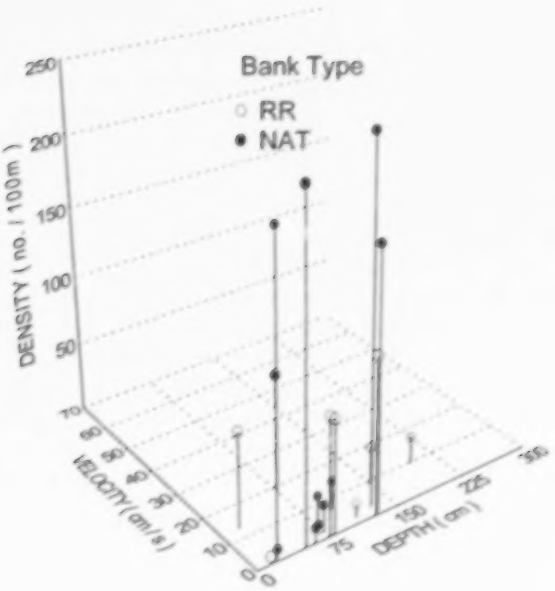
By the June survey, bank vegetation appeared to exert considerable influence on juvenile chinook habitat selection. Eighty–nine percent of natural bank sampling sites and 46% of rip-rap sites supported bank vegetation. River levels had risen 3.7 m on average between May and June study periods (Table 17). At these higher levels the river flooded into bank vegetation (Figure 13), resulting in extensive use of vegetative cover by juvenile chinook. Vegetation likely provided escape cover and velocity shelter, as evidenced by the positive relationship between chinook density and amount of bank vegetation at a site ($r^2 = 0.16$, $p = 0.01$, $df = 44$), and the negative correlation between density and nearshore velocity ($r^2 = 0.11$, $p = 0.02$, $df = 44$). Perhaps also a factor was the association of higher levels of vegetative cover with lower velocity ($r^2 = 0.16$, $p = 0.01$, $df = 44$). Low velocity sites, which tended to be natural banks, carried some exceptionally high chinook densities (Figure 14). Natural banks supported significantly higher mean chinook density than rip-rap in June (two sample $t_{44} = 2.93$, $p = 0.01$, $n = 46$), apparently due to greater vegetative cover and lower velocities (Table 20). At sites with a high vegetative cover rating, however, natural and rip-rap banks carried similar mean chinook densities (Table 21). On the other hand, natural banks with low vegetative cover supported significantly higher chinook densities (two-sample $t_{16} = 2.40$, $p = 0.03$, $n = 18$) than rip-rap banks in that category.

Table 20. Mean values of juvenile chinook density and selected habitat parameters at natural and rip-rap bank sampling sites during spring and summer surveys, lower Fraser River. Numbers in brackets represent 95% confidence intervals for estimates of mean density. Asterisks indicate statistically significant difference ($p < 0.05$) in chinook densities at natural and rip-rap banks.

Survey period	Chinook density (no./100 m)	Depth (cm)[a]	Velocity[a] (cm/s)	Vegetation[b] (%)
May				
- Natural banks (n = 11)	36.8 (10.5 – 122.6)	74.1	3.6	1.8
- Rip-rap banks (n = 8)	26.4 (6.4 – 100.2)	97.5	4.5	0
June				
- Natural banks (n=18)	91.7* (51.6 – 162.3)	130.6	8.1	61.1
- Rip-rap banks (n=28)	25.3* (12.2 – 51.4)	180.7	16.2	31.6
July				
- Natural banks (n = 20)	20.3 (8.6 – 46.2)	102.8	6.3	41.8
- Rip-rap banks (n = 30)	15.8 (8.6 – 28.2)	151.5	13.8	16.7
August				
- Natural banks (n = 6)	4.0 (0 – 37.5)	155.0	2.5	0
- Rip-rap banks (n = 8)	6.8 (1.7 – 21.9)	133.8	11.9	0

[a] Water depth and average water column velocity measured at 3 m from bank.
[b] Percent of bank length, within 3 m of shoreline, with overhanging brush or grasses.

(a)



(b)

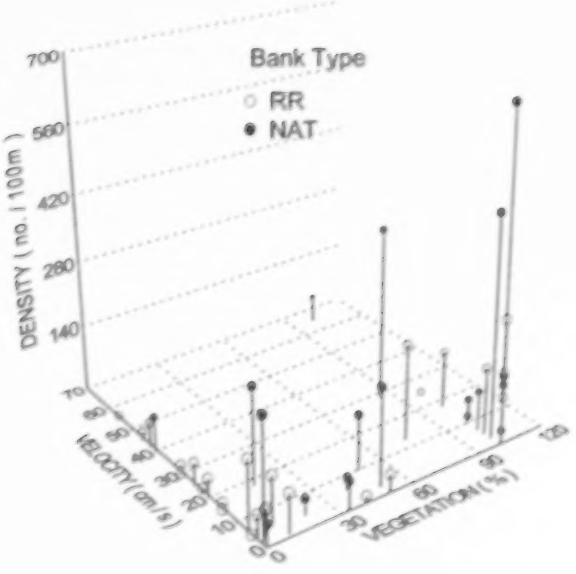


Figure 14. Juvenile chinook density at natural (NAT) and rip-rap (RR) banks in relation to selected habitat parameters during May (a) and June (b), lower Fraser River.

Table 21. Mean chinook density, vegetation and nearshore velocity at high and low cover sites by bank type in June and July, lower Fraser River. Cover ratings are based on percentage of site length with overhanging brush cover. Numbers in brackets represent 95% confidence intervals for mean density estimates. Asterisks indicate statistically significant difference ($p < 0.05$) in chinook densities at natural and rip-rap banks.

	June		July	
	Natural bank	Rip-rap bank	Natural bank	Rip-rap bank
High vegetative cover (>30%)				
No. of sites	13	15	8	5
Chinook density (no./100 m)	49.1 (44.5-190.9)	47.3 (17.8-123.5)	19.4 (5.7-61.9)	22.6 (0.6-344.9)
Vegetation (%)	79.2	56.4	78.5	88.0
Velocity (cm/s)	5.6	12.3	1.3	7.0
Low vegetative cover (0 – 30%)				
No. of sites	5	13	12	25
Chinook density (no./100 m)	89.8* (21.7-362.9)	12.0* (3.8-34.3)	20.9 (5.2-75.7)	14.7 (8.0-26.5)
Vegetation (%)	14.2	3.1	17.3	2.4
Velocity (cm/s)	14.6	20.6	9.6	13.2

During July, river levels had dropped approximately 2.2 m from the June peak (Table 17). While chinook densities at natural banks exceeded those at rip-rap by 28% on average (Table 20), the difference was not statistically significant (two-sample $t_{48} = 0.52$, $p = 0.61$, $n = 50$). Bank vegetation had less influence on chinook density at study sites in July ($r^2 = 0.03$, $p = 0.21$, $df = 48$), due to the drop in water level relative to the vegetation. Compared to June, a greater proportion of study sites fell in the low vegetative cover category (Figure 15 and Table 21). Further, natural and rip-rap banks supported comparable chinook densities, regardless of vegetative cover rating (Table 21). Chinook abundance was not significantly correlated, however, with any measured habitat parameter.

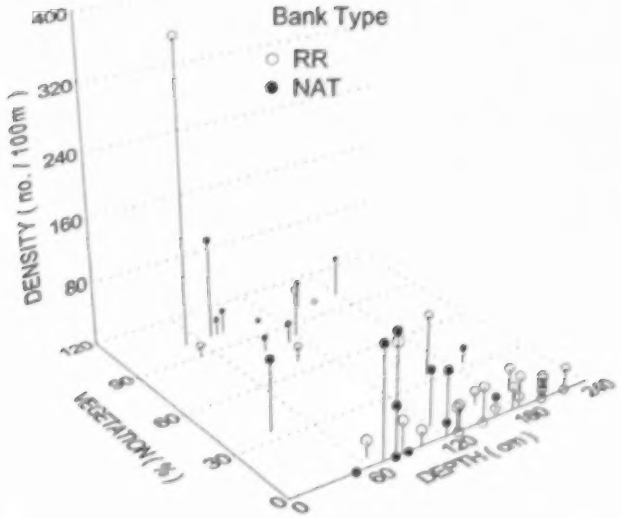
August field work was conducted at river levels approximately 4 m below those in June (Table 17). Vegetative cover was rated as zero at all study sites. A majority of natural (82%) and rip-rap (60%) sites supported no chinook juveniles. Juvenile chinook were absent from sampling sites in the lower study section, perhaps a reflection of lower nearshore velocities (mean 4 cm/s) relative to velocities in the upper study section (mean 11.7 cm/s). Analysis of data from 14 sampling sites in the upper study section indicated that juvenile chinook densities were positively related to nearshore velocity in August ($r^2 = 0.49$, $p = 0.01$, $df = 12$) but were not measurably influenced by water depth ($r^2 = 0.02$, $p = 0.56$, $df = 12$). There was no difference in juvenile chinook density at rip-rap and natural sites (two-sample $t_{12} = 0.50$, $p = 0.63$, $n = 14$).

Natural banks supported significantly higher chinook densities than rip-rap banks under high flow conditions in June (Table 20). However, there was no significant difference in relative chinook abundance at natural and rip-rap sites in July and August (ANOVA, $F_{2,104} = 2.03$, $p = 0.14$). Changes in relative density at natural and rip-rap sites between June and August may have been associated with an apparent shift in velocity preference, as implied by increasing chinook density at high velocity sites relative to low velocity sampling sites (Table 22). The significant change in relative densities between high and low velocity sites from June to August (ANOVA, $F_{2,104} = 3.34$, $p = 0.03$) may reflect a growth-related transition in habitat requirements (Lister and Genoe 1970; Everest and Chapman 1972).

Table 22. Seasonal change in relative abundance of juvenile chinook at high and low velocity sites, lower Fraser River, June-August. Numbers of sampling sites are shown in brackets.

Velocity category	Mean chinook density (no./100 m)		
	June	July	August
Low velocity (0 – 10 cm/s)	60.5 (30)	19.6 (28)	15.0 (9)
High velocity (> 10 cm/s)	20.1 (16)	15.1 (22)	17.0 (5)

(a)



(b)

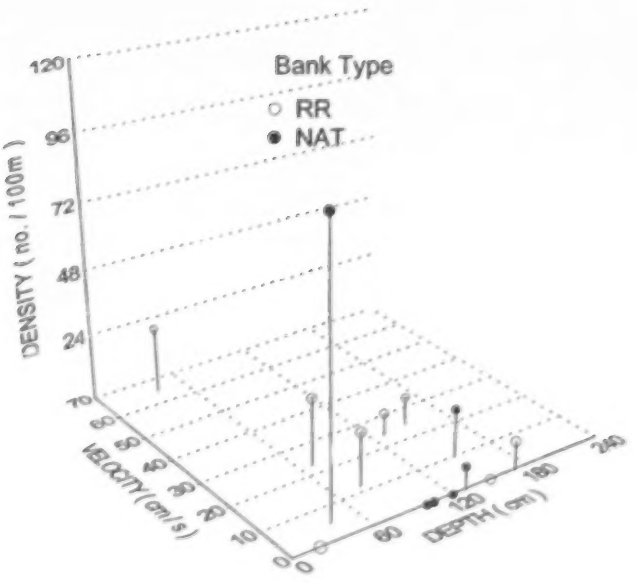


Figure 15. Juvenile chinook density at natural (NAT) and rip-rap (RR) banks in relation to selected habitat parameters in July (a) and August (b), lower Fraser River.

Winter Survey: Juvenile salmonid abundance was low and restricted to the upper study section (Hope to Agassiz) during winter. Salmonid juveniles in that section, including chinook and coho subyearlings and rainbow-steelhead parr (age 1+ and 2+), were present at 7 of 8 rip-rap sites but not any of 4 natural bank sampling sites (Table 23). The small number of juvenile salmonids captured (n = 21) indicated equal abundance of each species. This species composition was in contrast to May-August sampling when chinook juveniles comprised 97% of salmonids in the upper study section. Absence of over-wintering salmonid juveniles from the lower study section (Sumas River to Mission) was consistent with fish sampling results during the previous August.

Table 23. Fish sampling effort and numbers of natural and rip-rap bank sampling sites that supported over-wintering juvenile salmonids, lower Fraser River.

River section	Bank type	Sampling effort		No. of sites with salmonids present
		No. of sites	Bank length	
Hope to Agassiz	Natural[a]	4	65 m	0
	Rip-rap	8	183 m	7
Sumas River to Mission	Natural[a]	8	89 m	0
	Rip-rap	6	77 m	0

[a] Cobble-gravel and sand-silt banks.

Rip-rap sites in the upper section, where juvenile salmonids resided, were characterized by higher nearshore velocity and bank D_{50} than natural banks in that section, and both rip-rap and natural banks in the lower study section (Table 24). Correlation analysis, based on total juvenile salmonid captures, revealed salmonid density to be positively correlated with water velocity ($r^2 = 0.35$, $p = 0.04$, $df = 12$) and bank D_{50} ($r^2 = 0.43$, $p = 0.02$, $df = 12$) at sampling sites. The positive relationship between salmonid density and bank material size reflects the apparent preference of all species for the larger rock substrates at rip-rap sites.

At low river levels that prevail during winter, natural banks tended to have relatively small substrates that were unsuitable for salmonid over-wintering (Table 24). Some rip-rap banks were also unsuitable, because rock at the bank toe was embedded with fine silt and sand (Beniston et al. 1986). Rip-rap sites that were not embedded with silt at low winter flow levels are likely to have been high velocity situations under summer flows, and would therefore have provided relatively poor summer rearing habitat. Sites in the lower study section (Sumas River to Mission) exhibited significant bedload deposition in wood debris and rock substrates that would otherwise have provided over-wintering habitat .

Table 24. Mean values of juvenile salmonid density and measured habitat parameters at natural and rip-rap bank sampling sites in each study section, lower Fraser River, late winter.

Site characteristic	Upper section (Hope to Agassiz)		Lower section (Sumas River to Mission)	
	Natural banks	Rip-rap banks	Natural banks	Rip-rap banks
No. of sites	4	8	8	6
Chinook density (no./100 m)	0	1.2	0	0
Coho density (no./100 m)	0	1.4	0	0
Rainbow-steelhead parr density (no./100 m)	0	2.7	0	0
Velocity at 3 m (cm/s)	0	11.9	4.1	4.2
Depth at 3 m (cm)	155.0	133.8	72.5	173.3
Bank D_{50} (cm)	3.8	40.6	0.4	23.5

3.8 DISCUSSION

We compared steep, natural stream banks and steep banks altered by rip-rap protection with respect to their suitability for rearing and over-wintering of juvenile salmonids. In the North Thompson River and lower Fraser River studies, the effects of rip-rap on juvenile salmonid habitat were assessed by comparing fish utilization of unpaired rip-rap and natural bank sites. Though an attempt was made to consider effects of other physical factors, e.g. water velocity or depth, data analysis could not entirely account for the non-random distribution of rip-rap treatments. Along railway embankments, such as those on the North Thompson and Fraser rivers, rip-rap has been placed at sites where bank erosion had occurred. Such sites are relatively exposed to strong flows and subject to streambed scour. Rip-rap study sites therefore tended to have higher velocities and depths than the natural bank sites to which they were compared. Given the extreme variation in fish abundance data, it was not possible to completely isolate effects of physical circumstances, such as water depth and velocity, from effects of bank type (rip-rap or natural). Design of studies at the Thompson and Coldwater rivers, based on paired sampling sites with similar water depth or velocity, was intended to overcome this problem.

Study results suggest that rip-rap's effects on habitat suitability varied among study areas, fish species, life stages and seasons. Negative impacts of rip-rap were evident where habitat use by juvenile salmonids was strongly influenced by bank vegetation and associated physical features such as bank undercuts and wood debris accumulations. This type of situation applied to North Thompson River coho in late summer, lower Fraser River chinook at high flow in June, and Coldwater River chinook, coho and steelhead subyearlings at over-wintering habitat with high cover rating. These observed negative effects of rip-rap are consistent with the findings of other studies of anadromous salmonid juveniles, including ocean-type chinook juveniles in the Sacramento River (Schaffter et al. 1983; Michny and Hampton 1984) and Columbia River (Garland et al. 2002), and juvenile chinook, coho, cutthroat trout and steelhead in western Washington streams (Peters et al. 1998).

Studies reported here also indicated that the effects of rip-rap treatments could be neutral or positive in some circumstances. A neutral effect on juvenile salmonids was evident with respect to juvenile chinook and wild coho in the Coldwater River in late summer, chinook and rainbow trout parr in the Thompson River in early summer, chinook and coho in the North Thompson in winter, and chinook in the lower Fraser River at medium flow levels during May and August. Positive effects of rip-rap on salmonids were apparent in the Thompson River in late summer and winter, in the lower Fraser River in winter, and for steelhead parr in the Coldwater River during winter. Results of case studies described in this report indicate that a number of conditions influence rip-rap's effects on juvenile salmonid use of stream bank habitat. These circumstances are discussed in the following paragraphs.

3.8.1 Species, Life Stage and Season

Rip-rap bank treatments affected salmonid species and life stages differently. At the Coldwater River, for example, deep rip-rap sites supported significantly lower densities of over-wintering chinook and coho than deep natural bank sites with a high cover rating. While subyearling steelhead in the Coldwater River were also significantly more abundant at deep natural banks with high cover, steelhead parr densities were

significantly higher at deep rip-rap sites. At Thompson River, juvenile chinook and rainbow parr densities at large rip-rap exceeded those at natural cobble-boulder banks in late June and September, but density differences became more pronounced, and statistically significant, as the species grew and the season progressed. A marked seasonal shift in relative abundance of juvenile chinook at rip-rap in relation to natural banks, associated with a growth-related change in velocity preference, was also evident in the lower Fraser River between June and August.

Habitat selection by juvenile salmonids tends to become more defined in winter. While juvenile chinook and wild coho at Coldwater River exhibited no measurable preference for natural or rip-rap banks in summer, they clearly favoured deep, natural banks with abundant cover during winter. This pattern is consistent with findings of previous studies at Coldwater River (Swales and Levings 1986) and other coastal (Bustard and Narver 1975) and interior (Everest and Chapman 1972) locations.

3.8.2 Stream Size and Hydrology

A common characteristic of the large rivers in this study was the significant drop in water level between summer high flow and winter low flow periods. In the North Thompson River and lower Fraser River the influence of riparian vegetation and its associated cover features diminished as water levels receded away from the vegetated portion of the bank. Under winter low flows, availability of hiding cover near the bank toe, e.g., large rock (Hillman et al. 1987) or wood debris (Bustard and Narver 1975), determines habitat suitability for over-wintering. While vegetated, natural banks supported high summer densities of chinook or coho juveniles, it was rip-rap banks that carried greater numbers of these species at large river sampling sites during winter. Only rip-rap that remained unembedded by sand or silt deposition provided over-wintering habitat. Rip-rap banks suited for over-wintering were probably located at sites that were exposed to relatively high nearshore velocities under higher flow conditions during summer. At high flow stages, water velocities at such sites are likely to be excessive for summer utilization by juvenile chinook or coho. Rip-rap sites that are least attractive for summer rearing may therefore provide the most favourable conditions for over-wintering.

In the smaller Coldwater River, water level differences between late summer and winter did not account for substantial differences in habitat availability. For example, pools and deep glides that served as preferred summer rearing habitat also functioned as over-wintering habitat. Sites with relatively high water depths (>60 cm) and bank cover (>9%), whether natural or rip-rap, were preferred for both summer rearing and over-wintering.

3.8.3 Rip-rap Character

Larger rock increased habitat use by juvenile chinook and steelhead parr in the Thompson River. Effects of rock size were most apparent in comparisons between large rip-rap and natural cobble-boulder banks. These findings are at variance with a Columbia River study of subyearling ocean-type chinook (Garland et al. 2002). In that case, juvenile chinook tended to avoid rip-rap and preferred to rear at gradually sloping beaches of natural materials ranging from sand-silt to embedded boulders. One significant difference between the Thompson River and Columbia River situations is the

effect of nearshore water velocity. In the Columbia River, where velocities were reported to be low, juvenile chinook habitat selection was not measurably affected by differences in velocity. In the Thompson River, however, nearshore velocity differences, along with rip-rap size, were a major determinant of habitat suitability for chinook rearing (Lister et al. 1995).

It should be noted that rip-rap banks examined in our study had generally not been constructed in conformity with commonly used design criteria (e.g., U.S. Army Corps of Engineers 1969; RTAC 1975) which tend to promote a relatively smooth, hydraulically efficient surface along the rock blanket. At the time of this study, railway embankments on the large rivers were being protected from erosion by a maintenance program that frequently involved side casting of rock from railway cars. The rock was sized to meet site specific hydraulic conditions. This practice resulted in considerable site-to-site variation in rip-rap size and bank profile. Rip-rap banks at the Coldwater River were constructed of relatively large rock (mean D_{90} 99 cm) that was loosely placed to improve the bank's suitability for juvenile salmonid rearing. Large boulders (1 m - 1.5 m diameter) placed along the toe of some rip-rap banks supported increased juvenile chinook, coho and steelhead parr densities (Lister et al. 1995). These findings, and research on use of rip-rap by other fish species (Shields et al. 1995), suggest that the fish rearing capability of rip-rap banks can be improved by construction methods that achieve a rough, irregular profile.

3.8.4 Vegetation at Rip-rap Banks

A portion of rip-rap study sites on the North Thompson River (56%) and lower Fraser River (46%) supported bank vegetation which had, in all cases, resulted from natural re-vegetation. Rearing densities of juvenile coho (North Thompson) and chinook (lower Fraser River) juveniles were positively correlated with percentage of vegetated site length. At lower Fraser River study sites, rip-rap and natural banks with high vegetative cover rating (> 30% of site length) supported comparable densities of juvenile chinook at high summer flows in June and July. These observations suggest that in certain cases rip-rap sites can support vegetation, and that its presence has a positive effect on suitability of such sites for juvenile salmon rearing. The importance of vegetation is further emphasized at low vegetation cover sites (<30 % of site length) on the lower Fraser River, where just 14% of site length vegetation corresponded with a significant increase in chinook density (Table 21). Planting of woody vegetation is being used to increase the environmental value of rip-rap treatments (Sotir and Nunnally 1995), where sites only had to contain 30% or more vegetation to have high value. Many field practitioners may have assumed that this threshold would need to be greater.

3.9 ACKNOWLEDGEMENTS

Collection and initial analysis of data were funded by CN Rail and B.C. Ministry of Transportation. Fisheries and Oceans Canada funded report preparation. We are grateful to Luanne Patterson (CN Rail) and Michael Kent (Ministry of Transportation) for supporting this project.

3.10 REFERENCES

Beniston, R.J., D.B. Lister, and G.J. Naito. 1985. CN twin Track Project – Juvenile salmonid rearing studies, Thompson River, 1984-85. Prepared for CN Rail, Edmonton, Alberta.

Beniston, R.J., D.B. Lister, and G.J. Naito. 1986. CN Twin Track Project – Juvenile salmonid rearing studies, Fraser River, 1984-86, CNR Yale Subdivision Mile 40.1 – 94.5. Prepared for CN Rail, Edmonton, Alberta.

Beniston, R.J., W.E. Dunford, and D.B. Lister. 1988. Coldwater River juvenile salmonid monitoring study – Year 2 (1987-88). Prepared for B.C. Ministry of Transportation and Highways, Victoria.

Bloom, A.M. 1976. Evaluation of minnow traps for estimating populations of juvenile coho salmon and Dolly Varden. Prog. Fish Culturist 38: 90-101.

Bustard, D.R., and D.W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon and steelhead trout. J. Fish. Res. Bd. Canada 32: 667-680.

Chapman, D.W., and T.C. Bjornn. 1969. Distribution of salmonids in streams, with special reference to food and feeding. Pages 153-176 *In* T.G. Northcote (ed.) Symposium on salmon and trout in streams. H.R. MacMillan Lectures in Fisheries, University of British Columbia, Vancouver.

Everest, F.H., and D.W. Chapman. 1972. Habitat selection by juvenile chinook salmon and steelhead trout. J. Fish. Res. Bd. Canada 25: 1453-1464.

Fraser, F.J., P.J. Starr, and A.Y. Fedorenko. 1982. A review of the chinook and coho salmon of the Fraser River. Can. Tech. Rep. Fish. Aquat. Sci. 1126.

Garland, R.D., K.E. Tiffan, D.W. Rondorf, and L.O. Clark. 2002. Comparison of subyearling fall chinook salmon's use of rip-rap revetments and unaltered habitats in Lake Wallula of the Columbia River. N. Amer. J. Fish. Management 22: 1283 – 1289.

Gilbert, R.O. 1987. Statistical methods for environmental pollution monitoring. Van Nostrand Reinhold, New York.

Green, R.H. 1979. Sampling design and statistical methods for environmental biologists. John Wiley & Sons, Inc., New York.

Healey, M.C. 1991. Life history of chinook salmon. Pages 313-393 *In* C. Groot and L. Margolis (eds.) Pacific salmon life histories. UBC Press, Vancouver, B.C.

Hillman, T.W., J.S. Griffith, and W.S. Platts. 1987. Summer and winter habitat selection by juvenile chinook salmon in a highly sedimented Idaho stream. Trans. Amer. Fish. Soc. 116: 185-195.

Lister, D.B., and H.S. Genoe. 1970. Stream habitat utilization by chinook and coho salmon in the Big Qualicum River, B.C. J. Fish. Res. Bd. Canada 27: 1215-1224.

Lister, D.B., R.J. Beniston, R. Kellerhals, and M. Miles. 1995. Rock size affects juvenile salmonid use of streambank rip-rap. Pages 621-632 *In* C.R. Thorne, S.R. Abt, E.B.J. Barends, S.T. Maynord, and K.W. Pilarczyk (eds.) River, coastal and shoreline protection using rip-rap and armourstone. John Wiley & Sons Ltd., Chichester, England.

Michny, F., and M. Hampton. 1984. Sacramento River, Chico Landing to Redd Bluff Project – 1984 juvenile salmonid study. U.S Fish and Wildlife Service, Sacramento, California.

M. Miles and Associates Ltd. 1992. Coquihalla Highway: stability assessment of Coldwater River fisheries mitigation structures. Prepared for British Columbia Ministry of Transportation, Victoria.

Peters, R.J., B.R. Missildine, and D.L. Low. 1998. Seasonal fish densities near river banks stabilized with various stabilization methods. U.S. Fish and Wildlife Service, Lacey, Washington.

Platts, W.S. 1979. Relationships among stream order, fish populations, and geomorphology in an Idaho river drainage. Fisheries 4(2): 5-9.

Reynolds, J.B. 1981. Electrofishing. Pages 147-163 *In* L.A. Nielsen and D.L. Johnson (eds.) Fisheries Techniques. American Fisheries Society, Bethesda, Maryland.

RTAC (Road and Transportation Association of Canada). 1975. Guide to bridge hydraulics. C.R. Neill (ed.). University of Toronto Press, Toronto, Ontario.

Schaffter, R.G., P.A. Jones, and J.G. Karlton. 1983. Sacramento River and tributaries bank protection and erosion control investigation – evaluation of impacts on fisheries. California Dept. of Fish and Game, Sacramento.

Schmetterling, D.A., C.G. Clancy, and T.M. Brandt. 2001. Effects of rip-rap bank reinforcement on stream salmonids in the western United States. Fisheries 26(7): 6-13.

Schill, D.J., and J.S. Griffith. 1984. Use of underwater observations to estimate cutthroat trout abundance in the Yellowstone River. N. Amer. J. Fish. Management 4: 479 – 487.

Seber, G.A.F., and E.D. LeCren. 1967. Estimating population parameters from catches large relative to the population. J. Anim. Ecolcgy. 36: 631-643.

Shields, F.D., Jr., C.M. Cooper, and Samuel Testa III. 1995. Towards greener rip-rap: environmental considerations from microscale to macroscale. Pages 557 – 574 *In* C.R. Thorne, S.R. Abt, E.B.J. Barends, S.T. Maynord, and K.W. Pilarczyk (eds.) River, coastal and shoreline protection using rip-rap and armourstone. John Wiley & Sons Ltd., Chichester, England.

Sokal, R.R., and F.J. Rohlf. 1981. Biometry, 2nd edition, W.H. Freeman and Company, New York.

Sotir, R.B., and N.R. Nunnally. 1995. Use of rip-rap in soil bioengineering streambank protection. Pages 577-589 *In* C.R. Thorne, S.R. Abt, E.B.J. Barends, S.T. Maynord, and K.W. Pilarczyk (eds.) River, coastal and shoreline protection using rip-rap and armourstone. John Wiley & Sons Ltd., Chichester, England.

Swales, S., R.B. Lauzier, and C.D. Levings. 1986. Winter habitat preferences of juvenile salmonids in two interior rivers in British Columbia. Can. J. Zoology 64: 1506-1514.

U.S. Army Corps of Engineers. 1969. State of knowledge of channel stabilization in major alluvial rivers. Committee on Channel Stabilization. Tech. Rep. No. 7, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

WSC (Water Survey of Canada). 1980. Historical sediment data summary, Canadian rivers, 1978. Environment Canada, Ottawa.

WSC 1985. Surface water data, British Columbia, 1984. Environment Canada, Ottawa.

WSC 1986. Surface water data, British Columbia, 1985. Environment Canada, Ottawa.

WSC 1987. Surface water data, British Columbia, 1986. Environment Canada, Ottawa.

WSC 1988. Surface water data, British Columbia, 1987. Environment Canada, Ottawa.

WSC 1989a. Historical Streamflow Summary, British Columbia, to 1988. Environment Canada, Ottawa.

WSC 1989b. Surface water data, British Columbia, 1988. Environment Canada, Ottawa.

Whelen, M.A., and D.B. Lister 1985a. CN Twin Track Project – Juvenile salmonid rearing studies, North Thompson River, 1984. Prepared for CN Rail, Edmonton, Alberta.

Whelen, M.A., and D.B. Lister 1985b. Juvenile salmonid over-wintering study, North Thompson River, 1985. Prepared for CN Rail, Edmonton, Alberta.

Zallen, M., and J.Boyd. 1986. Investigations of habitat preferences by juvenile chinook salmon in the Fraser River mainstem, Floods-Hope area. Prepared for B.C. Ministry of Transportation and Highways, Victoria.

4.0 CONCLUSIONS AND RECOMMENDATIONS

4.1 CONCLUSIONS

The effects of rip-rap on fish and fish habitat fall into two broad categories: local effects and watershed-scale effects. Local effects of rip-rap on fish and fish habitat can be positive or negative depending on many factors. Watershed-scale effects such as constraints to natural fluvial processes are largely negative and can be cumulative in their impact. Although the results from both the literature review and case studies were varied, some trends emerged.

In systems that were considerably degraded with excessive erosion problems the application of rip-rap had positive effects on fish and fish habitat at a local scale (e.g. Thompson River case study). The definition of a degraded site can be somewhat subjective; however, results from the lower Fraser River suggested that natural sites with only 14% of their site length vegetated provided higher habitat value than rip-rapped sites. Further, the application of rip-rap to un-vegetated sites was most effective when combined with other habitat management strategies such as fencing, habitat complexing, and riparian planting. Positive effects from the application of rip-rap were most pronounced in steelhead parr density in our study and others (Knudsen and Dilley 1987; Swales et al. 1986.).

Rip-rap had negative effects on fish and fish habitat at a local scale in systems where natural sites were not severely degraded. As mentioned above, these natural sites were not 100% vegetated and pristine by any standard. Vegetated, natural banks were almost universally superior habitat than rip-rapped sites. The positive relationship between vegetation and fish density and the fact that rip-rapped sites with high vegetation had equivalent densities to natural sites highlight the importance of vegetation to juvenile salmonids. Schmetterling et al. (2001) noted that although rip-rap provides beneficial habitat to invertebrates and juvenile fish of certain species, the homogeneity of habitat does not support multiple age classes or species comparable to natural banks that include features such as vegetation and large woody debris. In terms of mitigation, the larger and more angular the rip-rap, the better. A bioengineering approach including incorporation of planting techniques and habitat complexing is preferable .

Although local effects of rip-rap appear mixed and dependent upon site condition (e.g. presence of vegetation), watershed-scale effects limit natural fluvial processes and disconnect rivers from their floodplains, decreasing ecosystem productivity and functionality. Watershed scale effects include restricted lateral channel migration, decreased natural sediment deposition, reduced recruitment of gravel and large woody debris, hydrological changes (such as reduced ability to attenuate flood peaks) and reduced heterogeneity of habitat. These effects are universally negative (Craig and Zale 2001) and potentially cumulative in nature (Stein and Ambrose 1998). Indeed, when numerous small decisions on linked environmental matters are made more or less independently, the combined consequences of the decisions are often not addressed (Stein and Ambrose 1998; Odum 1982). The effects from rip-rap should not be

considered solely in a site specific basis. Impacts to landscape-level ecological and hydrological processes should be considered during impact assessment stages.

4.2 RECOMMENDATIONS

4.2.1 Short term

1. DFO should be precautionary in the use of rip-rap, and only consider it as an option in degraded systems that are devoid of riparian vegetation and subject to excessive erosion.

2. DFO should encourage the use of alternative streambank stabilization methods, such as bio-engineering approaches (Adams 2003; MELP 2000), that may provide a greater value to fish habitat.

3. The application of rip-rap at vegetated sites will likely result in a loss of productive capacity and should be Authorized pursuant to Section 35(2) of the *Fisheries Act* provided the impacts from the proposed development cannot be fully mitigated.

4. If rip-rap is the only viable alternative for streambank protection, and is to be applied at a scale large enough to potentially affect river processes, DFO should require a hydrological assessment. The impact assessment should consider landscape-level ecological and hydrological processes, and downstream cumulative impacts to the watershed.

5. For Authorized rip-rap streambank protection projects, DFO should require compensation, including additional techniques other than simply re-vegetating the site. Consideration should be given to off-channel habitat development to compensate for the loss of potential for natural off-channel habitat development. Rigorous monitoring and maintenance programs should be implemented to ensure compensation success.

4.2.2 Long term

1. Dialogue with other levels of government and industry groups responsible for linear development (pipelines, railway, highways, etc.) should be initiated and maintained to develop strategies for protection of watersheds from excessive application of rip-rap as a bank protection technique.

2. Long term plans for river migration corridors should be developed. These plans should incorporate flood control solutions as alternatives to channelization. Examples include setback dykes at anticipated problem areas, stepped levees, greenbelts, meander zones, riparian conservation easements, subdivision regulations, building codes and zoning ordinances. A pro-active approach to

planning can increase the range of solutions available, and avoid dealing with projects on an emergency basis where potential solutions can be limited.

3. A coordinated education and awareness initiative should be undertaken targeting all private and industrial landowners and stakeholders within and adjacent to river migration corridors. The value of natural riparian vegetation in terms of both stream bank protection and fish habitat should be promoted to reduce unnecessary riparian removal and subsequent activities that result in an overall hardening of stream banks within a watershed without due consideration of its overall impact.

4. The effectiveness of alternate bio-engineering approaches to streambank protection should be quantified using a scientific monitoring and assessment program.

5. Long term field research should be conducted to fill the information gap on cumulative watershed level impacts resulting from extensive rip-rapping.

4.3 REFERENCES

Adams, M.A. 2003. Shoreline structures environmental design: a guide for structures along estuaries and large rivers. Fisheries and Oceans Canada. Vancouver, BC and Environment Canada, Delta, BC, 68p + appendices.

Craig, A.J., and A.V. Zale. 2001. Effects of bank stabilization structures on fish and their habitat - A literature review. US Geological Survey, Washington Department of Ecology, and Montana State University. Bozeman, MT, 29 pp.

DFO 1986. The Department of Fisheries and Oceans policy for the management of fish habitat. Ministry of Supply and Services, Canada., Ottawa. Cat No. Fs 23-98/1986E.

DFO 1998. Habitat conservation and protection guidelines. Second edition., Ottawa., 19pp. http://www.ncr.dfo.ca/publication_e.htm

Knudsen, E.E., and S.J. Dilley. 1987. Effects of riprap bank reinforcement on juvenile salmonids in four western Washington streams. N. Am. J. Fish. Manage. 7:351-356.

MELP 2000. Riprap design and construction guide. Public Safety Section, Water Management Branch. Victoria, BC, 65 pp.

Odum, W.E. 1982. Environmental degredation and the tyranny of small decisions. BioScience 32:728-729.

Schmetterling, D.A., C.G. Clancy, and T.M. Brandt. 2001. Effects of rip-rap bank reinforcement on stream salmonids in the western United States. Fisheries 26(7): 6-13.

Stein, E.D., and R.F. Ambrose.1998. Cumulative impacts of section 404 Clean Water Act permitting on the riparian habitat of the Santa Margarita, California watershed. *Wetlands* 18(3):393-408.

Swales, S., R.B. Lauzier, and C.D. Levings. 1986. Winter habitat preferences of juvenile salmonids in two interior rivers in British Columbia. Can. J. Zool. 64:1506-1514.